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STREAMBANK TAILINGS AND REVEGETATION STUDIES

STARS Phase III

FINAL REPORT

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Volume I

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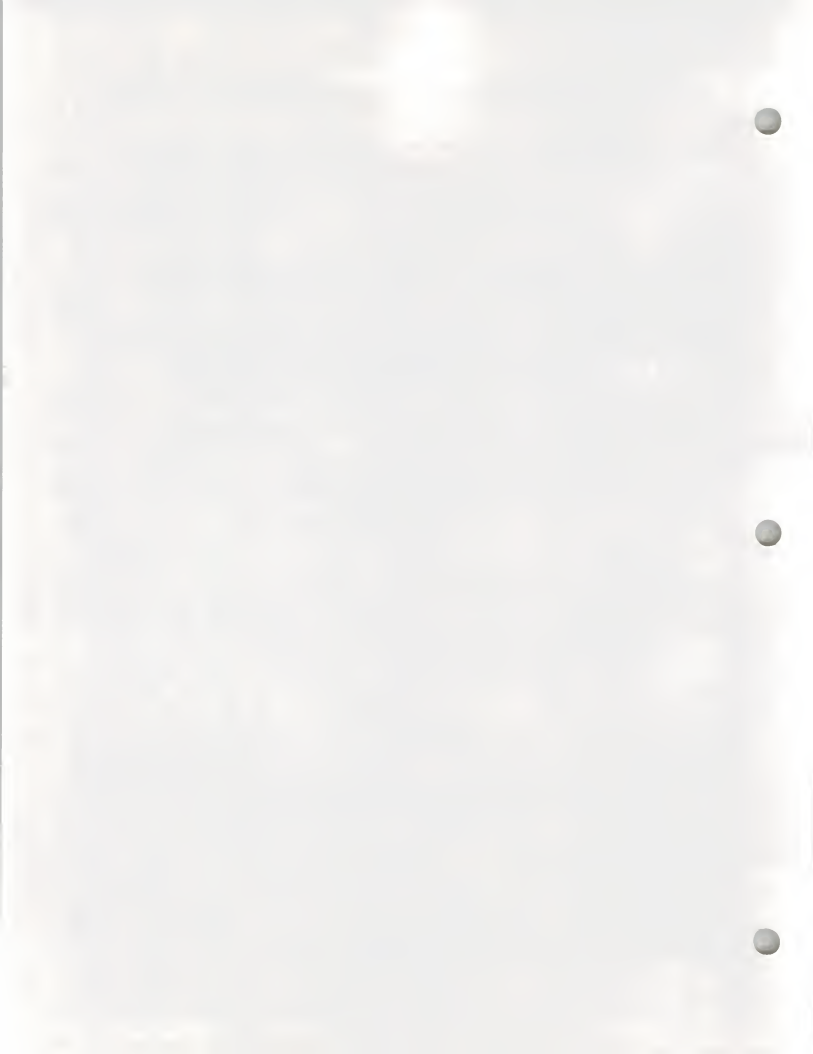
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FOREWORD

The STARS Phase III project has been completed through the joint efforts of the Reclamation Research Unit at Montana State University and Schafer and Associates, Inc., under the guidance of the Montana Department of Health and Environmental Sciences (MDHES) and reviewed by the Environmental Protection Agency (EPA). Early stages of the project were contracted through CH2M Hill with funds provided by EPA. Completion of the project has been accomplished by funding from Atlantic Richfield Company (ARCO), administered through MDHES.



1.0 EXECUTIVE SUMMARY

Over 100 years of nearly continuous mining activities have altered the natural environment surrounding the upper Clark Fork River. Early mining, milling, and smelting wastes were dumped directly into Silver Bow Creek, a tributary of the Clark Fork, and were subsequently transported downstream. Silver Bow Creek continued to receive raw mining and milling wastes until 1972, when a treatment plant was added to the Weed Concentrator in Butte. The mill tailings and other mining wastes in and near the creek contaminate downstream areas with potentially toxic elements: arsenic, cadmium, copper, lead, iron, and zinc.

Silver Bow Creek was listed as a Superfund site in 1983 by the United States Environmental Protection Agency (EPA) under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA). The site extends from Butte to Warm Springs, Montana, approximately 30 stream-miles. Potential methods to clean up the contamination of Silver Bow Creek are being studied under the Streamside Tailings Remedial Investigation/Feasibility Study (RI/FS), which is being prepared by the Atlantic Richfield Company (ARCO) with oversight provided by the Montana Department of Health and Environmental Sciences.

Approximately three to five million cubic yards of tailings are deposited within and along Silver Bow Creek. These tailings deposits pose a variety of actual or potential hazards to the environment, including acid production and movement of contaminated leachate to surface and ground waters; erosion and runoff to surface water of tailings and metal salts by direct precipitation and flood flows; erosion of tailings by wind action; toxicity to local plants and potential uptake of metals into the food chain of grazing animals; and potential risk to humans through direct contact and ingestion of contaminated soil and water. Remedial alternatives typically employed to ameliorate these hazards include removing the tailings to a secure storage area or capping them to reduce contact with air and water. Such techniques are commonly used for small areas or low volumes of wastes; however, for sites where contaminants cover large areas and involve large volumes of materials, the application of removal approaches may not be cost effective.

The Streambank Tailings and Revegetation Study (STARS) was conducted to develop and investigate the effectiveness of an innovative remedial technique to mitigate hazards associated with tailings along Silver Bow Creek. Since it would require at least five years to develop, construct, and monitor a new remedial technology, the STARS treatability study was begun prior to the Streamside Tailings RI/FS. The results of the STARS study will be considered in the evaluation of alternatives during the site RI/FS. The *in situ* remedial measures conceptualized by the STARS project team involved chemically neutralizing and fixing contaminants by incorporation of soil amendments into mine waste and revegetating with acid or metal-tolerant grass species. Soil amendments were specified to reduce the

mobility and bioavailability of most contaminants and vegetation cover would reduce the quantity and improve the quality of surface runoff and leachate passing through waste materials. Use of a soil cap of varying thickness in addition to soil amendments was also tested.

The STARS investigation was conducted in three phases. During Phase I, bench-scale soil-column and greenhouse studies were conducted. A variety of traditional and innovative chemical amendments were tested for effectiveness in neutralizing tailings acidity and reducing contaminant mobility. Metal- and drought-tolerant plant species were tested for their ability to survive and grow in amended wastes. In addition, tailings along Silver Bow Creek were classified and ranked based on physical and chemical parameters. Phase II initiated the pilot-scale field studies of the amendments and plant species found most effective during Phase I. One hundred field plots were established at five representative tailings sites along Silver Bow Creek. Final results of the Phase I and Phase II elements of the investigation were published in 1989. Phase III of the project consisted of three years of monitoring of the plots established during Phase II. This report describes the results of the Phase III of the STARS investigation.

Five sites representative of the physical and chemical conditions present in tailings and associated materials along Silver Bow Creek were investigated in the study. The five sites included three with fluvially-deposited tailings. One site had gravelly tailings (Rocker site), one sandy (Opportunity site), and one silty tailings (Ramsay Flats site). Two other sites, an upland agricultural field with a thin tailings layer (Fairmont site) and an historic tailings impoundment (Manganese Stockpile site), were also investigated.

Soil amendments consisting of combinations of limestone (CaCO_3) and hydrated lime (Ca(OH)_2), plus modest rates of ferric sulfate, and phosphogypsum were applied using a variety of mechanical incorporation techniques to four sets of replicated 16 by 20 foot plots at each site. Each plot received two different seed mixtures to evaluate the performance of a variety of plant species and each plot was fertilized. Five incorporation techniques were field tested including:

- Standard agricultural tillage to a depth of 15-20 cm (6-8 inches). A small tractor with a chisel plow, moldboard plow, and rototiller was used;
- Deep (1.2 m, 4 ft) mechanical plowing of amendments into tailings with a plow pulled by a D8-H Caterpillar bulldozer;
- Pressure-injection of an aqueous slurry of lime to a depth of 1.2 m (4 ft) into the waste materials;
- Standard agricultural tillage, followed by covering the amended tailings with a coversoil wedge that ranged in thickness from 0 to 46 cm (0-18 inches); and

- An experimental control which received no amendments, but was plowed using agricultural tillage, seeded and fertilized.

The STARS treatments were monitored for three years during the Phase III study. The data collected during the monitoring were used to evaluate the effectiveness of the treatments in eliminating or reducing the movement of metals from tailings to the surrounding environment. Both short-term and long-term effectiveness were investigated. The short-term effectiveness of the treatments was gauged through the direct measurement of soil and water physical and chemical parameters and certain vegetation parameters. The long-term effectiveness was extrapolated using standard modeling techniques to estimate surface water runoff, unsaturated flow through the vadose zone, and soil geochemical interactions. It is important to note that this study was designed to compare STARS treatments against untreated tailings conditions and to measure relative differences among treatments.

The general conclusions drawn from the STARS Phase III monitoring include the following:

1.1 SOIL HYDROLOGICAL MONITORING

- Unvegetated tailings at all sites have little capacity to store infiltrating rainfall or snowmelt. Up to three inches of recharge to groundwater is expected annually on untreated tailings.
- Coarse, granular tailings, such as those near the Rocker site, have the greatest potential for recharge to groundwater. Because of this, little runoff is expected from the coarse-textured tailings at Rocker, even from unamended tailings.
- Soil moisture can be removed by vegetation to depths as great as 3.5 feet. The establishment of vegetation on amended tailings decreased the potential for groundwater recharge, potentially enough to offset or prevent recharge entirely.
- Untreated tailings at all sites except Rocker have slow infiltration rates that lead to significant precipitation runoff with high concentrations of dissolved and total metals and arsenic.
- The coversoil, deep tillage, and agricultural treatments reduce runoff volume and improve runoff water quality significantly when compared to control plots.

- Soil moisture apparently migrated to groundwater at the Opportunity site in both the deep tillage and the control plots. Movement of water through the deep plow plots probably resulted from an improvement in infiltration caused by deep plowing.

1.2 SOIL CHEMISTRY MONITORING

- Treated plots generally show an increase in pore water pH, although the data are highly variable both areally and temporally.
- Native soils underlying tailings appears to provide buffering capacity and some protection from metals leaching to groundwater.
- Aluminum, iron, and copper become substantially less soluble in pore water with increasing pH. Manganese, cadmium, zinc, and lead become less soluble only at pH levels higher than 7.0.
- All treatments appeared to provide a reduction in contaminant leachability within the zone of amendment incorporation.
- Water soluble metals concentrations were dramatically reduced in the 0 to 15 cm depth interval in the treated plots at the Ramsay Flats and Opportunity.
- Water soluble arsenic concentrations increased notably in the 0 to 15 cm depth interval at Ramsay, but only slightly in the deeper intervals.
- Water soluble metals concentrations below the zones of incorporation were unchanged in all treatments, indicating that downward leaching of amendment materials (alkalinity) was negligible. This was also supported by the pore water chemistry data.
- All treatments exhibited some ability to reduce the phytotoxicity of the tailings materials.
- Extractable metals concentrations in the treated plots at Fairmont and Opportunity were generally lower than in the control plots.
- Measured pH (saturated paste) in soils at the 0 to 15 cm depth interval was higher than 7.0 in all treatments at all sites.
- The pH of soils was generally not altered below the treatment zone of incorporation.

- Total metals concentrations at the surface in most amended plots were similar to those in the control plots, except for the coversoil amendment, which had significantly lower surface metals concentrations.
- Deep tillage in thin tailings, such as at Fairmont, reduced surface metals concentrations, presumably by dilution through mixing with natural underlying materials.
- Both deep tillage and agricultural tillage produced acceptable results in amendment mixing, but the depth of incorporation was only 8 to 12 inches by agricultural tillage as compared to 3 to 4 feet by deep tillage.
- Amendment mixing by lime slurry injection was highly variable and generally not effective.

1.3 VEGETATION MONITORING

- Vegetation was established on all amended treatments except at the Manganese Stockpile site. Control plots had little or no vegetation at any site evaluated.
- After three years, vegetation rooting had progressed to a maximum depth of 15 inches in the amended plots and almost always followed zones of high pH and avoided zones of low pH.
- Optimum species mixes were determined for each of the four site types.
- Cadmium uptake by plants was evident for all treatments and may be a potential problem if amended areas are to be used for animal grazing.
- Vegetation failed at the Manganese Stockpile site. The failure was attributable to a high groundwater table, period flooding, and/or excessive soil salinity.

1.4 PREDICTIVE MODELING

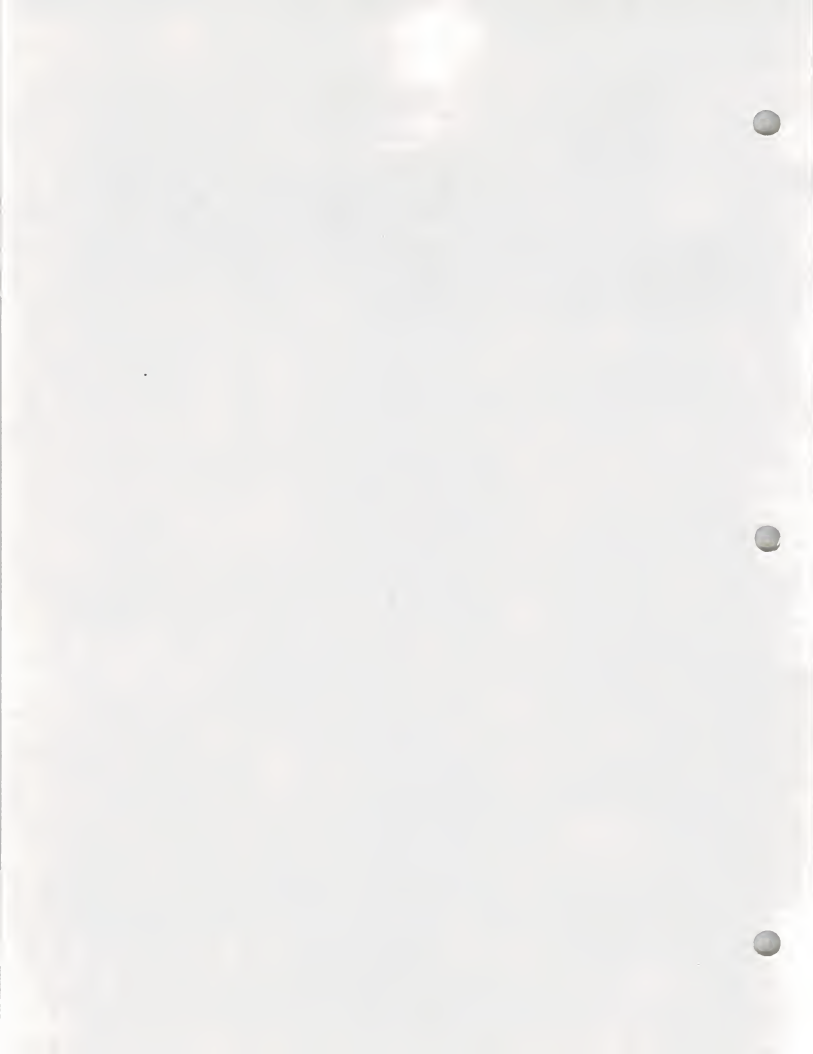
- The runoff/erosion model suggests that vegetated cover on treated plots will substantially reduce the volume of precipitation runoff and will limit the production of runoff to the late spring.
- Modeled erosion rates are reduced approximately by a factor of ten for the treated, vegetated plots when compared to untreated plots.

- The unsaturated zone flow model predicts that the establishment of vegetation on treated tailings would dramatically increase soil water use during the summer. Whereas approximately 3 to 5 inches of percolation to groundwater was predicted to occur annually on untreated plots, the establishment of vegetation on treated plots would eliminate percolation.
- The geochemical modeling predicts that the response of pH to lime addition is strongly nonlinear. Little lime was needed to increase the pH to 6.5, but 80 percent of the total limestone requirement was necessary to increase the pH from 6.5 to 7.5. Because of this, zinc and manganese control, which occurs at pH above 7.0, may be difficult through the use of limestone.
- Geochemical modeling showed that arsenic solubility would increase from one to three times as pH increases from 3 to 8. Arsenic solubility was assumed to be controlled by a combination of adsorption and precipitation.
- Geochemical modeling also indicates that precipitation of oxides and hydroxides is probably not the only process controlling metal solubility. Adsorption onto soil particles is probably another process involved in metals control.

1.5 OVERALL TREATMENT PERFORMANCE

- Agricultural tillage is easily implemented, provides good incorporation to a depth of 8 inches, and is the least expensive of the treatments tested.
- Deep tillage provides good incorporation to a depth of 3.5 to 4 feet, but requires more specialized, less maneuverable equipment than agricultural tillage. Deep tillage cost per acre is about twice that of agricultural tillage.
- Coversoil treatment was the most effective at establishing vegetative cover and eliminating runoff, and exhibited the lowest surface metals concentrations.
- The coversoil treatment and deep tillage produced similar vegetation production and cover. Agricultural tillage was less consistently effective than deep tillage in terms of vegetative measurements.
- Lime slurry injection was not an effective method of distributing lime within the zone of incorporation and could not be considered a cost-effective treatment.

In summary, it appears that STARS treatments have the potential to reduce the mobility and toxicity of metals *in situ*. Migration of metals as dissolved constituents in surface runoff and as eroded sediment was greatly reduced. In addition, the risk of contaminant migration into groundwater was reduced at all sites. Successful revegetation is expected to reduce exposure through other potential contaminant migration routes such as fugitive dust and direct contact. There are notable differences among the treatments studied, with no one treatment identified as applicable for treating the entire range of conditions found along Silver Bow Creek. Two primary amendment mixing methods, agricultural tillage and deep tillage, are probably the best-suited approaches for many parts of the site. The coversoil treatment will have applicability in certain areas, such as those where surface metals concentrations are high enough to cause concerns exist over existing or future human health exposure from direct contact.



2.0 INTRODUCTION

2.1 SITE HISTORY

Silver Bow Creek originates north of Butte, Montana, and is a major tributary to the upper Clark Fork River. The first recorded disturbance of Silver Bow Creek's natural channel occurred in 1864 when placer mining was initiated to extract low grade gold ore deposits along the creek and its tributaries (Freeman 1900; Meinzer 1914; Smith 1952). This mining activity was relatively short lived, with the cessation of most placer operations in the area by 1869, although minor placer activity continued on local streams.

Concurrent with placer mining along Silver Bow Creek, hard rock mining started on mineralized vein outcroppings on Butte Hill, north of Silver Bow Creek. Several small smelters/concentrators and a wet-process quartz mill, the Davis Mill, were built between 1866 and 1868 along Silver Bow Creek to process ore (Smith 1952). Although some copper and silver were being extracted, all facilities were closed by 1869. Apparently, little mining occurred in the Butte area from 1869 through 1874.

The first two mills in Butte were erected in 1874 to smelt gold and silver. William Farlin re-staked some mining claims on the Butte Hill in 1875 because of favorable assays of silver ore found in the area (Smith 1952). This rejuvenated mining activity in Butte; by 1878, several small smelters were operating in the area. Between 1879 and 1885, at least six major smelters were built along Silver Bow Creek from Meaderville to Williamsburg (Freeman 1900; Meinzer 1914; Smith 1952; Historical Research Associates (HRA) 1983). Silver was mined extensively in the 1870s, but by 1881, silver was less important as Butte became one of the nations prominent copper mining centers. A copper smelter (old works) was constructed near the mouth of Warm Springs Creek canyon at the new town of Anaconda, 27 miles west of Butte, in 1884 (Smith 1952).

The new Washoe smelter was constructed and opened on smelter hill, directly east of Anaconda, in 1903. The major smelters erected along Silver Bow Creek in the Butte vicinity operated nearly continuously until 1910 (HRA 1983). By 1910, Anaconda Copper Mining Company (ACM) had purchased and closed all but one of the major concentrators/smelters (the Pittsmont which operated until 1930) adjacent to Silver Bow Creek. Most of the ore was shipped via the Butte, Anaconda and Pacific Railway to the new Anaconda Smelter (Washoe) for processing.

By 1915, the Anaconda Copper Mining Company led the industry, a position it would hold for nearly 60 years. ACM commenced surface mining of low grade copper ore with the opening of the Berkeley Pit in 1955 and built the Weed Concentrator in 1963 to process this ore. In 1977, the Anaconda Copper Mining Company became a subsidiary of the Atlantic Richfield Company (ARCO). ARCO closed all underground mines in 1980 due to a depressed copper market and continued active mining only in the Berkeley Pit (established

in 1955). ARCO closed the Berkeley Pit in 1982 and the East Berkeley Pit in 1983, thus ending most active mining in Butte. Concurrently, the Washoe smelter in Anaconda was closed in 1980 and subsequently dismantled.

In 1986, Montana Resources Inc. (MRI) purchased Anaconda's Butte Operations (including the East Berkeley Pit, Continental East Pit, and Yankee Doodle Tailings Ponds). MRI resumed mining and is currently extracting copper ore from the East Berkeley Pit and processing mined ore in the Weed Concentrator. In 1989, the American Smelting and Refining Company (ASARCO) purchased 49 percent interest in MRI's Butte operation.

Over 100 years of continuous mining related activities have altered the area's natural environment. Early mining, milling, and smelting wastes were dumped directly into Silver Bow Creek and transported downstream. Silver Bow Creek continued to receive raw mining and milling wastes until 1972, when a treatment plant was added to the Weed Concentrator in Butte. Contamination problems were compounded by urban and domestic sewage, and by wastes from woodproducts treatment plants, phosphate and manganese production facilities, and chemical factories. Currently, there are no direct discharges to Silver Bow Creek from MRI's operations. The mill tailings and other mining wastes in and near the creek contaminate downstream areas with potentially toxic elements: arsenic, cadmium, copper, lead, iron, and zinc. These, and other elements were present in the mine ore and remain as by-products of the milling and smelting processes.

Silver Bow Creek and contiguous portions of the upper Clark Fork River were listed as a Superfund site in 1983 by the United States Environmental Protection Agency (EPA) under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA). The site presently extends from Butte to the Warm Springs Ponds.

2.2 SITE DESCRIPTION

Silver Bow Creek originates on the west slope of the continental divide approximately 5 miles north of Butte, Montana. The stream drains about 425 square miles at its confluence with Warm Springs Creek, which marks the beginning of the Clark Fork River. Elevations in the upper Clark Fork drainage range from 7000 to 8000 feet in headwater areas at the continental divide to about 3200 feet at Milltown.

Several Superfund sites have been established in the Clark Fork River Basin due to the complexity of the mine waste contamination. The Silver Bow Creek/Butte Area site extends from the uptown portion of Butte, Montana, downstream to the Warm Springs Ponds (Figure 2.1). Other Superfund sites in the area include the Montana Pole Site, the Anaconda Smelter Site, and the Milltown Reservoir Site. Montana Pole, also located in Butte, is a historic pole treating facility with PCP contamination. The Anaconda Smelter Site is centered at Anaconda, Montana 22 miles downstream of Butte. Most Butte ores were processed in Anaconda after 1904. The Milltown Reservoir Site, located near

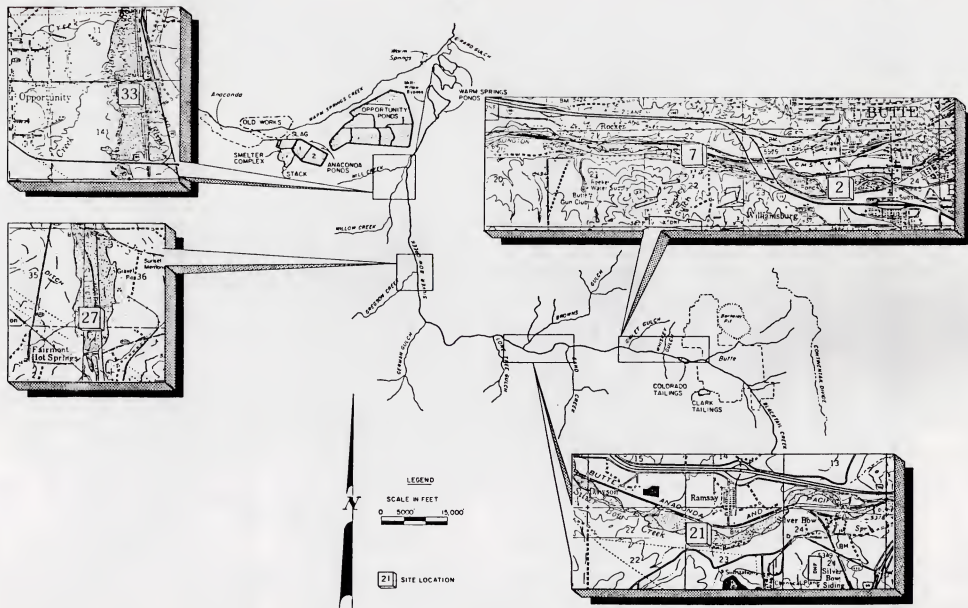


Figure 2.1 Silver Bow Creek CERCLA site between Butte and the Warm Springs ponds showing STARS Field locations.

Missoula, has several million yards of mine waste deposits in the reservoir which have contaminated ground water with arsenic and heavy metals. The Clark Fork River is also part of the Milltown Reservoir Site.

The Silver Bow Creek/Butte Area site includes the following operable units:

- Priority Soils (Butte Area);
- Mine Flooding (Butte Area);
- Nonpriority Soils (Butte Area);
- Active Mine Area (Butte Area);
- Streamside Tailings;
- Rocker Timber Treatment Plant; and
- Warm Springs Ponds.

All of the operable units, except Streamside Tailings, are under EPA lead. MDHES has the agency lead role for the Streamside Tailings Operable Unit. Each OU will proceed through the RI/FS process independently. As remedial action alternatives are developed during the FS, these alternatives will need to be evaluated not only for their effectiveness in mitigating problems with the OU, but also for their effect and relationship with the entire Silver Bow Creek/Butte Area site. The Streambank Tailings and Revegetation Study (STARS) is a treatability study which has been initiated to develop low cost, innovative remedial measures to potentially cleanup portions of the streambank tailings operable unit.

Air, soil, and surface and ground waters have been contaminated from tailings and other mining waste deposited in the Silver Bow Creek channel and floodplain. Several million cubic meters of mining wastes are in the floodplain of Silver Bow Creek. Remediation of these wastes is the focus of RI/FS activities for the Streambank Tailings Operable Unit. Streambank tailings are eroded by wind and water. Sulfide oxidation in the mine wastes increases acidity which increases the mobility of most metals. As water percolates through or flows across the barren streambank mine wastes, metals are leached into ground and surface waters. Water evaporating from the land surface leaves a residue of metal-enriched salts which can degrade air quality. These aspects are discussed in more detail in the following section.

2.3 CONCEPTUAL MODEL

Evaluating and selecting remedial options under CERCLA is influenced by the contaminants present at a site, their concentration, the volume of contaminated material, characterization of potential exposure pathways, and finally analysis of the risk of potential adverse effects from the contaminants on public health or the environment. The streamside tailings operable unit is complex due to the variety of mechanisms affecting the mobility of contaminants. To develop remedial methods for streambank tailings and to measure their effectiveness, potential mechanisms that expose humans or the environment to contaminants

in the streambank tailings must be understood. A conceptual model of the exposure pathways of contaminants in streambank tailings has been developed to show the fate and transport of streambank tailings contaminants (Figure 2.2).

The primary contaminants in streambank tailings are metals that are at elevated total or soluble concentrations, and that may adversely affect human health or the environment. These metals include Al, As, Sb, Cd, Cu, Fe, Mn, Hg, Ag, Pb, and Zn (Reclamation Research, Schafer and Associates, and CH₂M Hill 1989a). The metals are potentially very mobile. The potential mobility of each element is partly affected by chemical reactions occurring in tailings, and partly by release mechanisms that allow contaminants to be transported from the tailings (leachate, surface runoff, erosion, wind, and bioaccumulation; Table 2.1). The risk to human health and the environment from the site contaminants will be evaluated in a separate site risk assessment to be conducted by MDHES.

2.3.1 Soil/Solute Interaction

The primary chemical reaction in streambank tailings that affects contaminant mobility is acidification by oxidation of sulfide minerals (especially pyrite), and by hydrolysis of ferric iron to form ferric hydroxide ($\text{Fe}(\text{OH})_3$) (see Appendix D in the Phase II STARS Final Report; Schafer and Associates, Reclamation Research, and CH₂M Hill 1989b). Both of these reactions create acids, and in general the solubility of metals (Al, Cd, Cu, Fe, Mn, Ag, and Pb) increases as pH decreases. The mobility of As and Sb in soils tends to increase as pH increases, however. The mobility of Hg is influenced mainly by the solubility of Hg-organic complexes. Once contaminants are soluble, they are more readily mobilized by release mechanisms and transported out of the tailings. Hence the relative exposure to contaminants via this exposure route is strongly affected by soil chemical reactions within tailings.

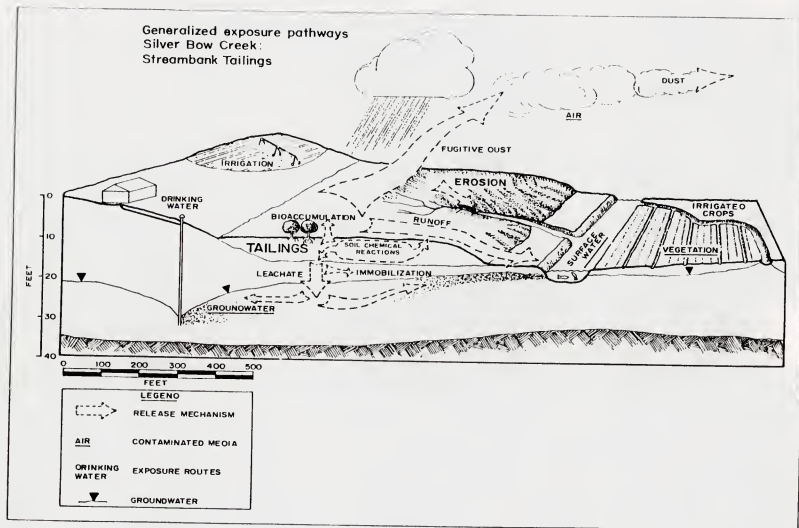


Figure 2.2 Typical exposure routes for contaminants in Silver Bow Creek streambank tailings.

Table 2.1 Contaminant exposure routes for the Silver Bow Creek streambank tailings.

SOURCE¹	RELEASE² MECHANISM	ENVIRONMENTAL³ TRANSPORT PROCESS	EXPOSURE ROUTE⁴ (POINT OF CONTACT)	OTHER⁵ MEDIA CONTAMINANTS
Contaminated Streambank Tailings (SOIL)	Soil Loss- Sediment	Surface Water	Ingestion (Drinking water, fish, aquatics, crops, livestock)	Soil
	Runoff- Dissolved Metals	Surface Water	Ingestion (Drinking water, fish, aquatics, crops)	Soil Vegetation
	Leachate	Ground Water	Ingestion (Wells-drinking water, crops, livestock)	Surface Water Vegetation
	Fugitive dust	Air	Inhalation (Breathing dust)	Soil
	Bio-accumulation	Vegetation	Ingestion (Crops, livestock)	none
	Direct Contact	Soil	Ingestion (Direct contact)	none

¹ - The source of contaminants are streambank tailings with elevated metal levels.

² - The release mechanism refers to the chemical or physical process releasing metal from the tailings.

³ - The transport media is the pathway metals are removed from the tailings.

⁴ - The exposure route (ingestion or inhalation) is the primary way humans are exposed to a contaminant, and the point of contact (in parentheses) is the medium inhaled or ingested.

⁵ - Contaminants can be exchanged between transport media.

2.3.2 Soil Loss

Tailings material can be moved physically into surface water or onto uncontaminated soil areas by wind and water erosion. The quantity of material moved depends on a variety of factors including vegetative cover, tailings texture, rainfall intensity and duration, slope angle, length of the tailings surface, and proximity to surface water. The quantity of contaminants moved is correlated to the gross erosion rate and to the total concentration of contaminants in the tailings. Establishing vegetative cover could decrease erosion by reducing the quantity of runoff and the tendency for sediments to be entrained by runoff.

2.3.3 Dissolved Metals in Surface Runoff

Water flowing across the tailings surface may solubilize metals and hence contribute to contaminant migration. This process, along with erosion, are probably the main contributors to metal transport. Dissolved metal levels in surface runoff will be influenced by the quantity of surface runoff (affected by infiltration rate of tailings), and the solubility of metals in surface tailings layers (see 2.3.1). The untreated tailings are barren of vegetation, and as a result water evaporates from the tailings leaving surface crust of soluble metal-enriched salts. Preventing this formation of surface salt crusts and increasing infiltration rate could dramatically reduce migration of dissolved metals in surface runoff.

2.3.4 Leachate

Water can move downward through tailings and exit the base of the tailings zone. Soil chemical reactions in tailings strongly affect metal concentrations in the leachate. Earlier project documents (Phase I RI report, Multitech 1987) reported that soil layers buried by tailings have highly elevated metal levels suggesting the importance of leachate as a release mechanism. Samples of pore water from below the tailings/soil interface collected as part of the 1986 supplementary RI data (CH2M Hill 1987) also had high metal levels.

The migration of metals in leachate to groundwater is affected by the quantity of leachate, by the solubility of metals in tailings, and by chemical reactions of metals which occur in the tailings, the tailings/soil interface, and the groundwater surface (or capillary fringe). Where the groundwater is several feet below the tailings, a considerable (but undetermined) fraction of the dissolved metals in leachate may be immobilized by chemical reactions. However, the degree of immobilization may vary by element (Cd and As are more mobile than Pb). In addition, characteristics of the underlying soil or sediment may also affect immobilization. Finally, the amount of water draining through the tailings zone can also be influenced by changes in water use by vegetation.

2.3.5 Fugitive Dust

Tailings deposits are often susceptible to wind erosion because slopes are flat and unprotected, and the tailings cover large areas. High velocity winds that entrain small soil particles may occur in portions of the project area. This fugitive dust can be inhaled by humans living in areas where it is transported. In addition, fugitive dust can be redeposited on the surface of uncontaminated soils where it increases the volume and area of contamination and where it is also susceptible to erosion. Dust deposited on snowpack may runoff into surface water. Dust may also adhere to forage and accumulate in grazing livestock or wildlife. Establishing vegetative cover will significantly reduce fugitive dust transport.

2.3.6 Bioaccumulation

The danger of bioaccumulation of metals from the untreated tailings is minimal because little currently grows on the tailings. Bioaccumulation is mostly a concern on treated tailings. Movement of metals through the food chain can be extremely complex. The tendency for plants to accumulate metals in aboveground biomass varies for each contaminant. The accumulation of metals by grazing animals depends on plant uptake and on animals ingesting soil (which is influenced by range condition, and the animals grazing habits). Although accumulation of metals by humans is equally complex, some criteria have been established to limit the dietary intake of certain elements in various food. Changes in soil chemical reactions (especially by liming) can affect the rate of plant uptake of metals. In addition, different plant species have differing tendencies to accumulate metals.

2.3.7 Direct Contact

Direct contact is an exposure route easily understood and has straightforward control measures. Quantifying the actual exposure, the exposed population, and the dosage may be difficult, however. The risk to human health from direct contact is by ingesting soil material. Whereas, deliberate consumption of soil may be rare (pica), unintentional ingestion may occur by contact with soiled hands, inhaling dust from soiled clothes, and inhaling/ingesting soil by recreational users of tailings and other contaminated areas. Establishing vegetative cover, or capping tailings with coversoil may be an effective means to reduce direct contact.

2.4 DESCRIPTION OF STARS INVESTIGATION

2.4.1 Background

Effective remedial measures for streambank tailings must solve a variety of problems. These include acid production and movement of contaminated leachate to surface and ground waters, erosion by water and wind, and contact hazard to humans, livestock, and wildlife. Typical remedial alternatives, which may ameliorate most of these problems, include removing to a secure storage area or capping to reduce infiltration and air entrainment. These techniques work for small areas or low volumes; however, for sites where contaminants cover large areas and involve large volumes of materials, they may not be applied cost-effectively. Since streambank tailings along Silver Bow Creek are disseminated over much of its 44 kilometer (27 mile) length and involve millions of cubic meters of material, the Montana Department of Health and Environmental Sciences (MDHES) recognized that new and innovative technologies should be investigated as remedial alternatives. One of these technologies is chemically modifying *in situ* streambank tailings characteristics (pH, plant-available metals) followed by revegetating with adapted plant species.

2.4.2 STARS Treatability Studies

An innovative remedial measure conceptualized by the project team involved 1) chemically neutralizing and fixing contaminants by deep incorporation of soil amendments into mine waste, 2) possibly using a soil cap (at least in selected areas), and 3) revegetating with acid or metal-tolerant species of grasses. Soil amendments would reduce the mobility and bioavailability of most contaminants whereas the soil cap (if used) and vegetation cover would reduce the quantity of leachate passing through waste materials.

The Streambank Tailings and Revegetation Study (STARS) was initiated to develop an innovative remedial technique and to investigate the effectiveness of that technique. The STARS project was divided into three components.

- Phase I: Bench-scale soil column/greenhouse treatability studies designed to develop and test amendments.
- Phase II: Pilot-scale treatability studies which included implementing in the field the remedial options designed in Phase I.
- Phase III: Field monitoring program to evaluate the response of the wastes to the amendments. This response has been evaluated on the basis of reduced leachate quantity, abating metal flux to surface and soil waters, vegetation productivity and persistence, and changes in soil chemistry.

Stars Phase I: Objectives of Phase I were to 1) test a variety of chemical amendments for their effectiveness in neutralizing acidity and in reducing contaminant mobility, 2) evaluate drought and/or acid-tolerant plant species for their ability to grow in amended waste, and 3) develop a ranking system for classifying waste materials based on easily identifiable physiochemical characteristics.

A statistical waste ranking model (Schafer and Associates et al. 1989) was developed so that samples from streambank tailings could be analyzed in the field for basic chemical and physical properties and could then be classified into a defined waste class. A specific remedial measure could then be developed for each waste class, thus simplifying the design of the remediation effort. As a result of this ranking model, three chemically and physically distinct classes of streambank tailings were identified. Three other significant waste classes identified were impounded tailings, flood-affected natural soils, and soils buried by tailings.

Representative samples of waste were collected from the streambank tailings area to test the validity of the ranking model and to provide test material for the column and greenhouse studies. Chemical amendments selected for testing were those with 1) acid-

neutralizing capacity, 2) available phosphorus, 3) soluble calcium, 4) decomposable organic matter, and 5) other fixation agents (zeolites, and ferric sulfate).

The column tests evaluated the short-term effect of soil amendments on leachate metal concentration. In general, when a combination of chemical amendments including CaCO_3 , Ca(OH)_2 and ferric sulfate were added to waste samples, the mobility of most metals was reduced by one or more orders of magnitude. The combination of chemical amendments which best reduced metal mobility were proposed for use in greenhouse studies and in subsequent Phase II STARS field trials.

Plant species were selected for testing in the greenhouse based primarily on 1) their tolerance to drought, low pH, high EC, and high metal levels, and 2) their commercial availability. The performance of these species were evaluated when grown on unamended and chemically-treated wastes. Plant performance was evaluated based on survival under stress conditions, total yield, root growth, and metal accumulation. Seed mixtures consisting of suitable plant species were prescribed for use in Phase II field tests. Details of Phase I activities are described in:

- Final Summary Report - Silver Bow Creek RI/FS, STARS, Phase I: Bench-scale Soil Column and Greenhouse Treatability Studies, and Tailings Ranking System. Prepared by Reclamation Research Unit, Schafer and Associates, and CH2M Hill. SBC-STARS-Phase I-F-R1-102589. October 25, 1989.

Stars Phase II: Objectives of Phase II were to implement in field the amendments that were most effective in neutralizing tailings acidity and in reducing metal content in the laboratory leachate solution, and that demonstrated the best plant establishment, growth and survival in the greenhouse.

Five sites were selected for testing the different amendment incorporation methods and plant species. Site selection was based on the physicochemical characteristics of the original samples and by using the tailings ranking systems. The surface material on each of the five sites was characterized in order to locate the experimental field plots. To identify the most uniform physicochemically areas in which to establish the field plots, samples were collected in a grid pattern. These samples were analyzed for pH, electrical conductivity (EC), and metal concentration by X-ray fluorescence spectrometry (XRF). These data were kriged and isoline concentration maps prepared that identified areas that were the most homogenous. This information facilitated the placement of the field plots.

Blocks of five or six treatments were implemented at each site. These treatment blocks were replicated four times at each site to facilitate statistical comparisons.

Once the plot locations were identified and staked, they were sampled to determine the key soil properties. Analysis of sulfur fractions, nutrient levels, and basic soil properties (pH, EC and texture) were used to develop individual lime and fertilizer rates for each

study plot. Results of individual calculations are presented in detail in the STARS Phase II Final Field Plot Construction and Vegetation Seeding Document (Schafer and Associates et al. 1989). Secondary amendments (ferric sulfate and phosphogypsum) were added to some study plots to further reduce metal mobility through chemical precipitation of insoluble compounds and through adsorption.

A second series of field experiments were conducted in small (1 x 1 m) microplots to determine the rate at which pH reached equilibrium (between 7.5 and 8.5) in the field, and to determine if any of the plots needed to be artificially hydrated to achieve pH equilibrium. These experiments were conducted to verify the Phase I bench-scale laboratory findings and to insure that the results were proper for use in the field. Results of the microplot studies were used to finalize the field plot amendment rates.

Five amendment incorporation techniques were implemented in the field during Phase II. Treatment 1 involved standard agricultural tillage to a depth of 15-20 cm (6-8 inches). A small tractor with a chisel plow, moldboard plow, and rototill was used. Treatment 2 involved deep (1.2 m, 4 ft) mechanical plowing of amendments into tailings with a plow pulled by a D8-H Caterpillar. For Treatment 3, lime slurry was pressure injected 1.2 m (4 ft). Treatment 4 involved incorporating amendments by standard agricultural tillage, followed by constructing a coversoil wedge ranging from 0 to 46 cm (0-18 inches) of material. Treatment 5 was a control which received no amendments, but the control plots were prepared by tillage with standard agricultural equipment, fertilized, seeded, and mulched.

A least-cost blend of fertilizer material was formulated for each study plot and consisted of ammonium nitrate, di-ammonium phosphate and potassium chloride. Fertilizer boron was added to the mixed fertilizer for each plot as borax.

Seeding recommendations for each of the five sites were based on the greenhouse evaluation of plant performance in Phase I. Two seed mixtures were recommended for each site. Seeds of recommended species for each mixture were weighed and placed in individual packages for each study plot. These packages were kept in cold storage until seeding.

Seeding was originally proposed for late fall 1988, however, by November many plots still had pH values greater than 8.5. Hence, seeding was delayed until early spring 1989. Each plot was divided in half and the different mixture applied to each side. Plots were seeded in April and May 1989.

For a thorough discussion on the rationale of Phase II activities consult:

- Final Summary Report - STARS Phase II: Field Scale Treatability Study Plot Construction. Prepared by Schafer & Associates, Reclamation Research Unit, and CH2M Hill. SBC-STARS-PhaseII-F-R1-051789. May 17, 1989.

2.4.3 Project Documents

In addition to the aforementioned documents, several other STARS documents were prepared to guide Phases I, II, and III of the STARS project. These documents should be reviewed to fully understand the entire project.

- Draft Work Plan for the Silver Bow Creek Streambank Tailings and Revegetation Studies. Phase I. Prepared by Schafer & Associates and Reclamation Research Unit. May 14, 1987.
- Draft Applicable, Relevant, or Appropriate Requirements (ARARS) for the Silver Bow Creek Streambank Tailings and Revegetation Studies (SBC-STARS). Reclamation Research Unit, and Schafer & Associates. 1987. September 7, 1987.
- Quality Assurance Project Plan (QAPP) for the Streambank Tailings and Revegetation Studies. Silver Bow Creek RI/FS. EPA Region VIII. Prepared by Reclamation Research Unit and Schafer & Associates. Document No. SBC-STARS-QAPP-F-R0-121187. December 11, 1987.
- Greenhouse Studies Operation Plan (GSOP) for the Streambank Tailings and Revegetation Studies (STARS). Silver Bow Creek RI/FS. EPA Region VIII. Prepared by Reclamation Research Unit and Schafer & Associates. Document No. SBC-STARS-GSOP-F-R1-121187. December 11, 1987.
- Tailings Studies Operation Plan (TSOP) for the Streambank Tailings and Revegetation Studies (STARS). Silver Bow Creek RI/FS. EPA Region VIII. Prepared by Schafer & Associates and Reclamation Research Unit. Document No. SBC-STARS-TSOP-F-R1-121187. December 11, 1987.
- Laboratory Analytical Protocol (LAP) for the Streambank Tailings and Revegetation Studies. Silver Bow Creek RI/FS. EPA Region VIII. Prepared by Reclamation Research Unit and Schafer & Associates. Document No. SBC-STARS-LAP-F-R1-121187. December 11, 1987.
- STARS - Phase II Pilot Scale Treatability Study Work Plan. Streambank Tailings and Revegetation Study. Field Plot Construction and Vegetation Seeding. Document No. :SBC-STARS-II-PSTS-F-R1-071288. Prepared by Schafer & Associates and Reclamation Research Unit. July 12, 1988.
- Final Technical Memorandum - Silver Bow Creek RI/FS. STARS: Tailings Ranking & Data Analysis of Silver Bow Creek Waste. Prepared by Schafer

and Associates, Reclamation Research Unit, and CH2M Hill. February 10, 1989.

- STARS - Transition Phase Technical Memorandum for the Streambank Tailings and Revegetation Studies. Methods for Conducting Field Site Selection and Sampling of Phase II STARS plots. Silver Bow Creek RI/FS. EPA Region VIII. Prepared by Reclamation Research Unit and Schafer and Associates. SBC-STARS-SFSSS D-R0-042988. June 10, 1988.
- Technical Memorandum - STARS Phase III. Runoff, erosion and leachate prediction using HELP and CREAMS. Prepared by Schafer and Associates, and CH2M Hill. October 16, 1989.
- PHASE III: Final Work Plan for Monitoring Treatability Studies. Silver Bow Creek RI/FS. EPA Region VIII. Prepared by Schafer & Associates, Reclamation Research Unit and CH2M HILL. Document No. SBC-STARS-PhaseIII-WP-F-R0-110189. November 01, 1989.
- Sampling and Analysis Plan, Part I: Quality Assurance Project Plan (QAPP). STARS: PHASE III. Monitoring of Field Treatability Plots. Silver Bow Creek RI/FS. EPA Region VIII. Prepared by Reclamation Research Unit, Schafer & Associates and CH2M HILL. Document No. SBC-STARS-III-SAP(QAPP)-F-R0-110189. November 01, 1989.
- Sampling and Analysis Plan, Part II: Field Sampling Plan (FSP). STARS: PHASE III. Monitoring of Field Treatability Plots. Silver Bow Creek RI/FS. EPA Region VIII. Prepared by Schafer & Associates, Reclamation Research Unit and CH2M HILL. Document No. SBC-STARS-III-SAP(FSP)-F-R0-110189. November 01, 1989.
- Technical Memorandum - Vegetation Response in the STARS field plots-1989. Silver Bow Creek RI/FS. STARS Phase III. Prepared by Reclamation Research Unit, Montana State University. Document No. SBC-STARS-III-VEG-RES-D-R1-122289. December 29, 1989.
- STARS PHASE III Work Plan Addendum. Silver Bow Creek RI/FS. EPA Region VIII. Prepared by Schafer & Associates, Reclamation Research Unit and CH2M HILL. Document No. SBC-STARS-PhaseIII-WPA-D-R0-030191. March 01, 1991.
- Activity Report. Streambank Tailings and Revegetation Studies: STARS Phase III. Silver Bow Creek RI/FS. Prepared by Reclamation Research Unit, and Schafer & Associates. Prepared for CH2M HILL. October 23, 1990.

- Narrative Report. Streambank Tailings and Revegetation Studies. STARS Phase III. Silver Bow Creek RI/FS. Prepared by Reclamation Research Unit, and Schafer & Associates. Prepared for Montana Department of Health and Environmental Sciences. October 17, 1991.
- Technical Memorandum - Mortality of Vegetation on STARS Field Plots at the Manganese Stockpile Site. STARS Phase III. Silver Bow Creek RI/FS. Prepared by Reclamation Research Unit, Montana State University. 1992.



3.0 STARS PHASE III PROGRAM OBJECTIVES

The primary purpose of the STARS Phase III monitoring program was to measure the effectiveness of STARS treatments to abate contaminant movement and reduce the phytotoxicity of the tailings. This was accomplished by 1) measuring soil chemical, physical, and hydrologic properties, and evaluating plant performance at each site; and then 2) comparing data collected for each treatment to determine relative differences in measured properties. Treatment effectiveness will be inferred in this document from these data. Specific data collection tasks were:

- Measure soil chemical properties and identify differences between treatments that may affect contaminant solubility and bioavailability, or that may affect long-term treatment effectiveness;
- Evaluate erosion (soil loss) and runoff rate;
- Measure dissolved and total metal concentrations in surface runoff;
- Measure metal cation and anion concentrations in soil pore water;
- Measure metal loadings in forage plants; and
- Measure the survival and performance of seeded vegetation.

3.1 MONITORING APPROACH

The objectives outlined above were addressed by measuring response variables of the STARS plots in three basic categories; the soil hydrology; the soil chemistry; and the vegetation. Long-term effectiveness of treatments, and fugitive dust contributions are assessed by interpreting or modeling data collected in these three categories.

Monitoring of soil hydrologic response included metal transport by soil loss, dissolved metal movement in runoff, and groundwater recharge by leachate. A critical requirement to assess each of these elements of hydrologic response was to develop a water budget for each STARS treatment and for unaltered tailings.

Soil chemical changes through time and the effectiveness of STARS treatments in altering chemical conditions of streambank tailings were assessed by repeated soil core monitoring at various depths in the STARS plots.

Finally, vegetation monitoring and sampling was used to evaluate short- and long-term revegetation success, and to evaluate the potential for bioaccumulation of metals in and on forage.

3.1.1 Phase II Experimental Design

The implementation of several amendment incorporation techniques was described in STARS Phase II. Blocks of five or six treatments were implemented at five different locations within the Operable Unit. Waste materials at each site represented the different types of tailings found along Silver Bow Creek. The treatment blocks were replicated four times at each site to provide for statistical comparisons of the response variables. Table 3.1 exhibits amendment treatments used at each site. The Manganese Stockpile (STARS Site 2) was located within the Butte Reduction Works, east of the present day Butte Sewage Treatment Plant. STARS Site 7 was located adjacent to Silver Bow Creek near the town of Rocker. The large expanse of flood deposited wastes near the town of Ramsay was the location of STARS Site 21 (Ramsay Flats). The fourth STARS site (27) was situated north of Fairmont near the "Yellow Ditch", while the last site (33) was located adjacent to Silver Bow Creek near the town of Opportunity. Figures 3.1 through 3.5 show the design of the blocks, replications, and treatment plots.

Table 3.1. Soil amendment treatments used in the STARS field sites.

Site	Soil Amendments ¹	
	Main Plots	Extra Plots ²
2 Manganese Stockpile Site	CaCO ₃ , Ca(OH) ₂ , phosphogypsum, and ferric sulfate	Ca(OH) ₂
7 Rocker Site	CaCO ₃ , Ca(OH) ₂	None
21 Ramsay Flats Site	CaCO ₃ , Ca(OH) ₂ , phosphogypsum, and ferric sulfate	None
27 Fairmont Site	CaCO ₃ , Ca(OH) ₂ , and ferric sulfate	CaCO ₃ , Ca(OH) ₂ and TSP
33 Opportunity Site	CaCO ₃ , Ca(OH) ₂ , and phosphogypsum	None

¹ Phosphogypsum is a by-product of fertilizer production. TSP is a fertilizer, triple superphosphate (0-45-0).

² The soil amendments applied to the 'Extra' plots were incorporated using agricultural tillage treatment.

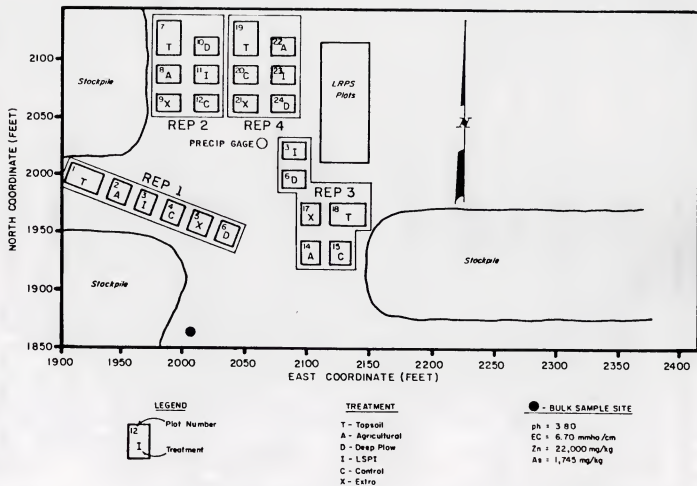


Figure 3.1. Detailed location of STARS Phase II field plots and treatments for the Manganese Stockpile, Site 2.

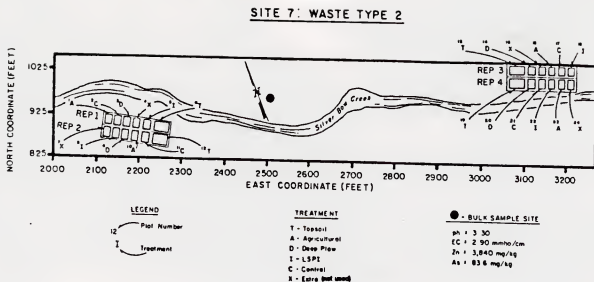


Figure 3.2. Detailed location of STARS Phase II field plots and treatments for the Rocker Site, Site 7.

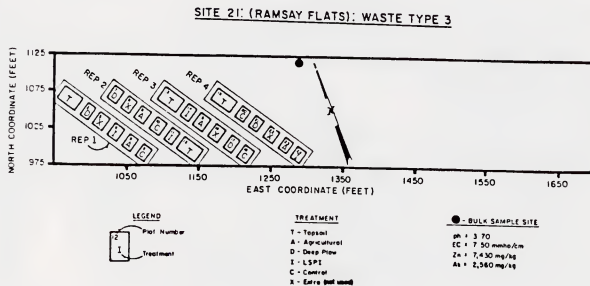


Figure 3.3. Detailed location of STARS Phase II field plots and treatments for the Ramsay Flats, Site 21.

SITE 27: TYPE F - FLOODED AGRICULTURAL SOIL

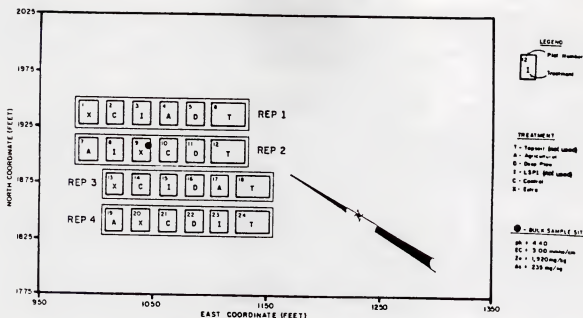


Figure 3.4. Detailed location of STARS Phase II field plots and treatments for the Fairmont Site, Site 27.

SITE 33 TYPE 1 WASTE

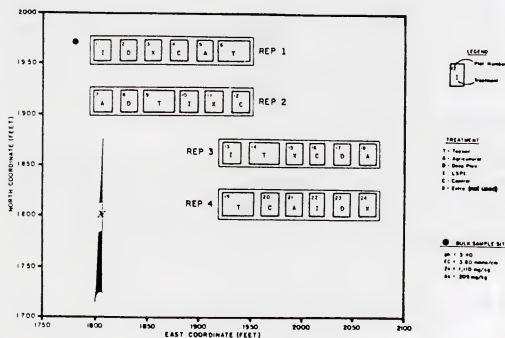


Figure 3.5. Detailed location of STARS Phase II field plots and treatments for the Opportunity Site, Site 33.

3.1.2 Incorporation Techniques From STARS Phase II

The soil amendments selected from STARS Phase I (Table 3.1) were incorporated into the waste materials using a variety of techniques. The techniques were chosen based on depth of amendment incorporation, commercial availability, and cost of the incorporation method.

Five incorporation techniques were field tested in Phase II. Treatment 1 involved standard agricultural tillage to a depth of 15-20 cm (6-8 inches). A small tractor with a chisel plow, moldboard plow, and rototiller was used. Treatment 2 involved deep (1.2 m, 4 ft) mechanical plowing of amendments into tailings with a plow pulled by a D8-H Caterpillar. For Treatment 3, a aqueous slurry of lime was pressure injected 1.2 m (4 ft) into the waste materials. Treatment 4 involved incorporating amendments by standard agricultural tillage, followed by constructing a coversoil wedge ranging from 0 to 46 cm (0-18 inches) of material. Treatment 5 was a control which received no amendments. After all amendments were incorporated, water was applied and rototilled into the plots to encourage the hydration of lime and reaction with the waste materials. Fertilizers, determined by site characteristics, were applied. The fertilizer was rototilled into the plots which prepared the surface for seeding.

Seeding recommendations for each of the five sites were based on the greenhouse evaluation of plant performance in Phase I. Two seed mixtures were recommended for each site. Seeds of recommended species for each mixture were weighed and placed in individual packages for each study plot. Sites 7, 21, 27 and 33 were seeded April 12-14, 1989. Site 2 was too wet to seed during this period and was seeded on May 19, 1989 (see Section 4.3). A detailed description of the construction of the experimental plots and implementation of the treatments is provided in the Phase II Final Summary Report (Schafer & Associates, Reclamation Research Unit, and CH2M Hill, Inc. 1989).

3.2 MONITORING FREQUENCY

The monitoring program had three basic elements: 1) soil hydrological monitoring, 2) soil chemical monitoring, and 3) vegetation monitoring. The frequency of monitoring for each of these elements is shown in Table 3.2. The appendix location of data for each part of the monitoring program is also shown in the table. The monitoring program of each element is described in the following sections.

Table 3.2. Organization and frequency of Phase III data collection and data appendix location.

Monitoring Category	Year				Report Appendix
	1989	1990	1991	1992	
----- Frequency -----					
<u>Soil Hydrology</u>					
Rainfall/Runoff Testing				1	A2, B4
Metals/Anions in Soil Pore Water		1	1	1	B3
Soil Water Content		5	8	9	A2
Soil Pore Water pH/SC/Eh		5	8	9	B3
Depth to Groundwater		5	5	9	A2
Climate	C ¹	C	C	C	A1
<u>Soil Chemistry</u>					
Ph, SC and Soluble Metals		1			B1, B3
Ph, SC and Bio-Available Metals			1		B2, B3
Surface Metals				1	B5
Amendment Mixing				1	C4
<u>Vegetation Monitoring</u>					
Seedling Density	1				D1
Areal Cover	1	1	1		D2, D3, D4
Production			1		D5
Rooting Characteristics			1		D6
Vegetation Metals			1		D7

¹ Begins in October 1988, ends in August 1992.

3.3 SOIL HYDROLOGICAL MONITORING

3.3.1 Water Budget

The cornerstone of soil hydrologic monitoring is developing a water budget for each STARS treatment and for untreated streambank tailings. Development of a water budget will quantify the effects of STARS treatments on infiltration, runoff, leachate generation, and plant water use. Changes in infiltration will affect the movement of metals through surface runoff, while changes in the quantity of leachate generated may affect movement of metals into groundwater. Once a water budget is established for each STARS treatment, models can be used to estimate the long-term impact on metal movement by surface runoff or leachate pathways (see Section 5.0 for discussion).

A water budget is a mass balance of water within the soil profile (Equation [3.1]). Inputs of water to the soil zone (from precipitation or capillary flux from groundwater) are balanced by losses (drainage, runoff, or evapotranspiration) and by changes in stored soil water. Methods are available to either measure or estimate all components of the water budget.

$$\text{PPT} = \pm \text{SW} + \text{RO} + \text{ET} \pm \text{DRAINAGE} \quad [3.1]$$

where PPT is total precipitation (measured)
ET is evapotranspiration (calculated from PET)
RO is runoff (calculated from runoff studies)
SW is the net change in soil water storage (measured)

Precipitation was measured at all sites and continuously at Ramsay Flats (Site 21). Evapotranspiration (ET) was estimated by calculating potential ET from climatic data and then comparing observed water use (as determined by changes in soil water content) to potential ET. Changes in stored soil water were determined by measuring soil water content periodically with a neutron probe. Runoff was calculated as a function of rainfall intensity during rainfall simulation tests in 1992.

Climatic data required for calculating the water budget included continuous precipitation measurements, maximum and minimum daily temperatures (for calculating PET), and daily solar radiation (also for calculating PET). Climatic data were collected from a station constructed at Ramsay Flats (Site 21). The climate station was linked to a field data acquisition system to facilitate continuous data retrieval. Data were retrieved at monthly intervals. Simple storage rain gauges were installed at other sites so that significant differences in precipitation among sites could be measured. Records of rainfall, maximum and minimum temperature, solar radiation, and PET were assembled for each station (Appendix A1).

3.3.2 Soil Water Content

Soil water content was measured using a neutron probe. This method measures water content by placing a source that emits high-speed neutrons (usually 50 mCi of Am-Be) at a selected depth in the soil profile. Thin-wall steel tubes just slightly larger in diameter than the probe were installed in one set of replicated STARS plots at each site for a total of 25 tubes (Table 3.3). Tubes were installed to a depth of 2.4 to 3.0 m (eight to ten feet) so that most tubes encroached into the saturated groundwater zone. During installation samples were collected from each zone of differing lithology or texture to determine field water gravimetric content (See Appendix A2 for data). After access tubes were installed, the holes were logged with a neutron probe and a gamma-source which measures soil wet bulk density. Since the volumetric water content (as measured by the neutron probe) is equal to the gravimetric water content times the dry bulk density and the gamma density probe measured wet bulk density, a computer based mathematical iteration was used to

Table 3.3. Locations of monitoring installations.

Site	Plot Number	Treatment Code ¹	Monitoring Instrumentation				Depth of Tailings (cm) ⁴
			Neutron Access Tube	Piezometer	----- Pore Water Sampler ----- Number	Depth of Installation (cm)	
2	19	T ¹	✓	✓	2	40 and 90	79.5
	20	C	✓	✓	3	40, 90, and 150	114
	21	X	✓	✓	2	40 and 90	126
	22	A	✓	✓	2	40 and 90	102
	23	I	✓	✓	3	40, 90, and 150	117
	24	D	✓	✓	3	40, 90, and 150	-- ³
7	13	T	✓	✓	2	40 and 90	-- ⁵
	14	D	✓	✓	3	40, 90, and 150	--
	16	A	✓	✓	2	40 and 90	42-48
	17	C	✓	✓	3	40, 90, and 150	--
	18	I	✓	✓	3	40, 90, and 150	--
21	13	T	✓	✓	2	40 and 90	36
	14	I	✓	✓	3	40, 90, and 150	63
	15	A	✓	✓	2	40 and 90	39
	17	D	✓	✓	3	40, 90, and 150	--
	18	C	✓	✓	3	40, 90, and 150	34
27	13	X	✓	✓	2	40 and 90	-- ⁷
	14	C	✓	✓	3	40, 90, and 150	--
	16	D	✓	✓	3	40, 90, and 150	--
	17	A	✓	✓	2	40 and 90	--
33	7	A	✓	✓	2	40 and 90	35.5
	8	D	✓	PNI ²	3	40, 90, and 150	--
	9	T	✓	PNI	2	40 and 90	16
	10	I	✓	✓	3	40, 90, and 150	32
	12	C	✓	✓	3	40, 90, and 150	35

¹ T = Coversoil Wedge; C = Control; X = Extra; A = Agricultural; I = Slurry Injection; and D = Deep Plow.

² PNI = Piezometers were not installed in these two experimental plots.

³ Installation depths for lysimeters on coversoil plots was referenced to the original tailings surface.

⁴ Depth determined from neutron probe core logs.

⁵ Due to deep plow mixing, no depth of tailings was determined for these plots.

⁶ Tailings at Site 7 were well mixed with natural alluvium and were not usually discretely observable.

⁷ No tailings discernable in cores from Site 27.

obtain the highest correlation coefficient. The program began using the factory calibration for determining the water content which was then used to calculate the dry bulk density. It should be emphasized that no matter how accurate the calibrations are, or indeed, if any calibration was done, the relative change in water content between measurement dates will still be valid. Further, if plant water extraction is unimportant (such as on bare tailings), an increase in water content of deeper layers in the soil profile implies that overlying layers have reached and exceeded their field capacity, thus allowing downward percolation of infiltrated water. Calibration curves were then constructed relating neutron counts to volumetric water content (Appendix A2).

Selected samples collected while installing the neutron probes also had a soil water content curve developed from desorption data. It should be noted that since the neutron probe access tubes were installed following plot treatments, soil properties within the treated zones are those of the soil plus amendments and any soil structure changes produced by amendment incorporation. To construct this curve, an undisturbed core sample was taken to the laboratory and water content measured after the sample has equilibrated to known soil water potential levels. Soil water content was measured at several soil water potential levels. These desorption data aided in interpreting the soil water content data by indicating the direction of water flux for specific water content ranges (Appendix A2).

Neutron probe data served two purposes in the STARS Phase III monitoring program. Neutron tubes were monitored regularly during the growing season and during the rest of the year to construct a water budget for each STARS treatment. It should be noted that the water budget (Equation 3.1) and neutron probe data will evaluate drainage whether it is positive or negative and thus, determine if any upward movement of groundwater occurs in response to ET demands and/or capillary action. These factors could affect the effective longevity of applied amendments. In addition, neutron data were collected more frequently on selected plots during the simulated rainfall/runoff tests. Some neutron probe data exhibit anomalies due to reading errors when an interval was read twice or skipped. Where these errors were obvious, corrections were made. A second problem occurred during March 1992 in which the probe cable malfunctioned and was replaced with one with improperly marked intervals. These data are presented, but represent interval in error by approximately 7.5 cm.

Small scale rainfall simulation was used to collect response from the STARS plots. A highly modified Meeuwig rainfall simulator was used to simulate runoff from a rainstorm of 3.81 to 5.08 cm/hr (1.5-2 inches/hr). The simulations were conducted on the instrumented plots at Sites 7, 21, and 33. Response variables measured were rainfall rates, quantity of runoff, pH, SC, total and dissolved metal concentrations, anion levels, and total soluble solids. These rainfall/runoff test data are found in Appendices A2 and B4. The rainfall/runoff relationships were incorporated into the rainfall/runoff model calibrated for each site.

Samples of pore water were obtained from the unsaturated zone with ceramic-tip pressure-vacuum type pore water samplers (also known as suction lysimeters). Pore water samplers were installed at selected depths (Table 3.3) in sets of treated plots at each site corresponding to the depth of amended tailings, amended natural soil material, and the zone below the depth of incorporation. Lithologic cross sections of the instrumented replication at each of the STARS sites are exhibited in Figures 3.6a through 3.6e. A schematic of a typical pore water sampler installation is shown in Figure 3.7. The actual number of samplers varied depending on the specific treatments installed at each site (Table 3.3). Pore water chemistry results are used in conjunction with water data from the water budget indicating the quantity of drainage from each STARS treatment to develop an estimate of metal movement out of the tailings as leachate.

Pore water pH, SC, and Eh levels were measured on five occasions in 1990, on eight occasions in 1991, and on nine occasions in 1992. The data are exhibited in Appendix B3.

Concentrations of total metals, dissolved metals and cations, and anions were determined in the soil pore water sampler in 1990, 1991, and 1992 (Appendix B3).

3.4 SOIL CHEMICAL MONITORING

Soil chemical analysis were used to measure the short-term effectiveness of various STARS treatments in neutralizing acidity, to measure the bioavailability of metals in each STARS treatment, and to measure the relative mobility of metals.

Five soil chemistry monitoring activities were performed. Each is identified in Table 3.4 along with the purpose for collecting the data, the frequency of collection, and the appendix locations of the data.

Soil pH, EC levels, and concentrations of soluble metals were measured in saturated paste extract solutions prepared from amended tailings/soils collected from various depths within the profile. Profiles were collected from each experimental plot. Concentrations of bio-available metals were determined in extracts of ammonium acetate (pH 7.0), while bio-available arsenic levels were measured in Bray P-1 extracts of the amended tailings/soils.

Surface materials (0-2.5 cm) from each plot were collected from each STARS experimental plot including the control plots. Surface materials outside, but adjacent to each replication, were also collected. These samples represent pre-treatment and the no action alternative. The collected samples were analyzed for total metal concentrations (As, Cd, Cu, Pb, and Zn) by XRF spectroscopy.

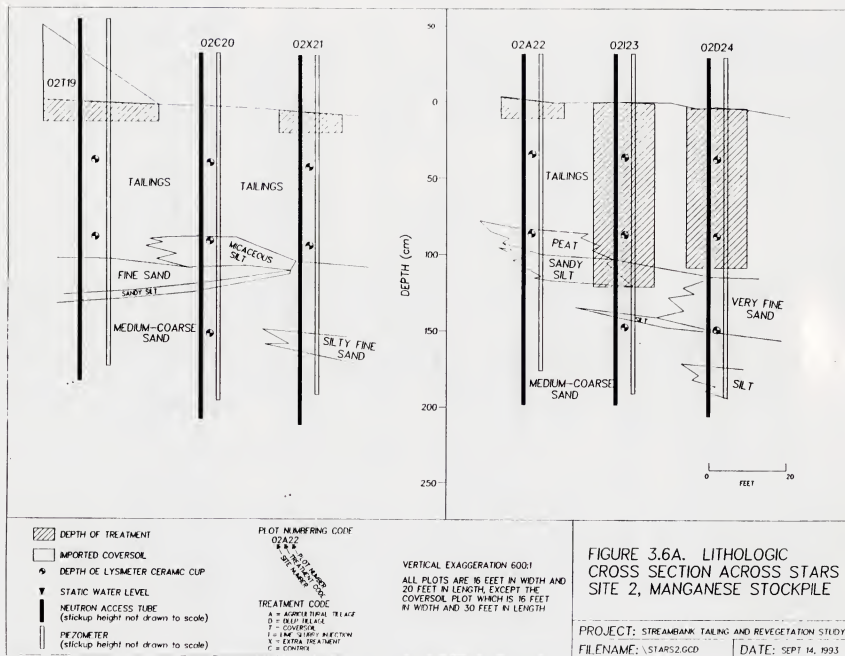


Figure 3.6a. Lithologic cross section of instrumented replication, STARS Site 2 (Manganese Stockpile).

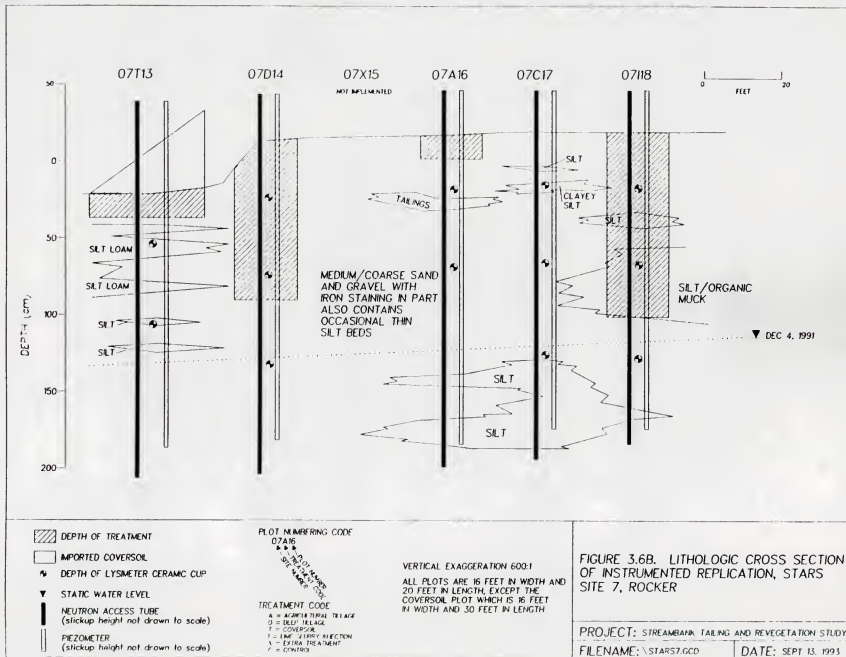


Figure 3.6b. Lithologic cross section of instrumented replication, STARS Site 7 (Rocker).

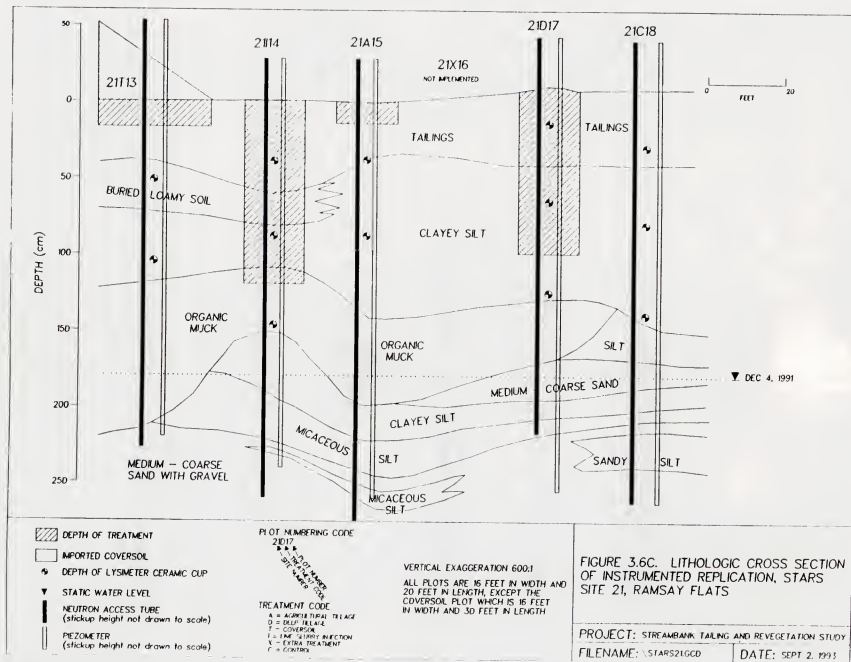


Figure 3.6c. Lithologic cross section of instrumented replication, STARS Site 21 (Ramsay Flats).

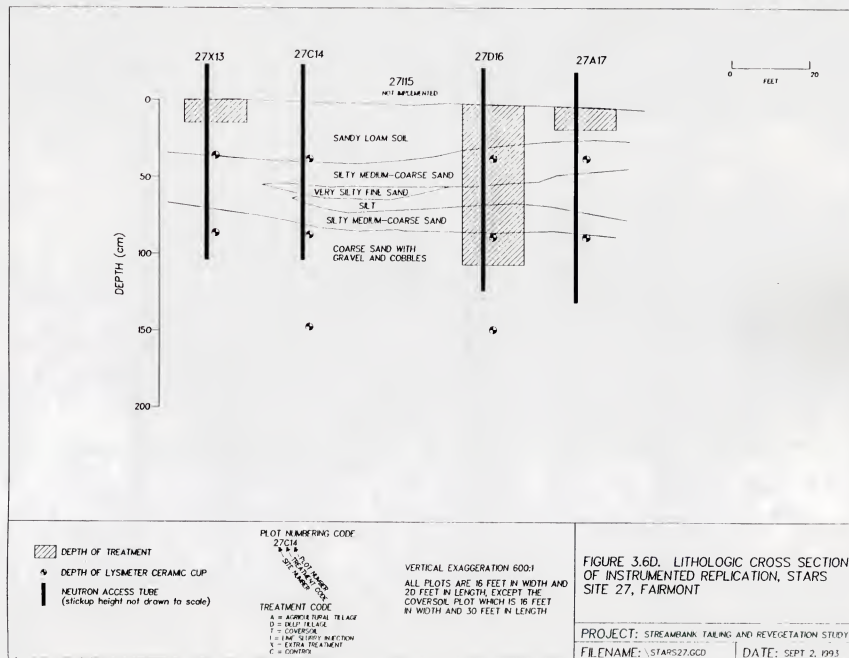


Figure 3.6d. Lithologic cross section of instrumented replication, STARS Site 27 (Fairmont).

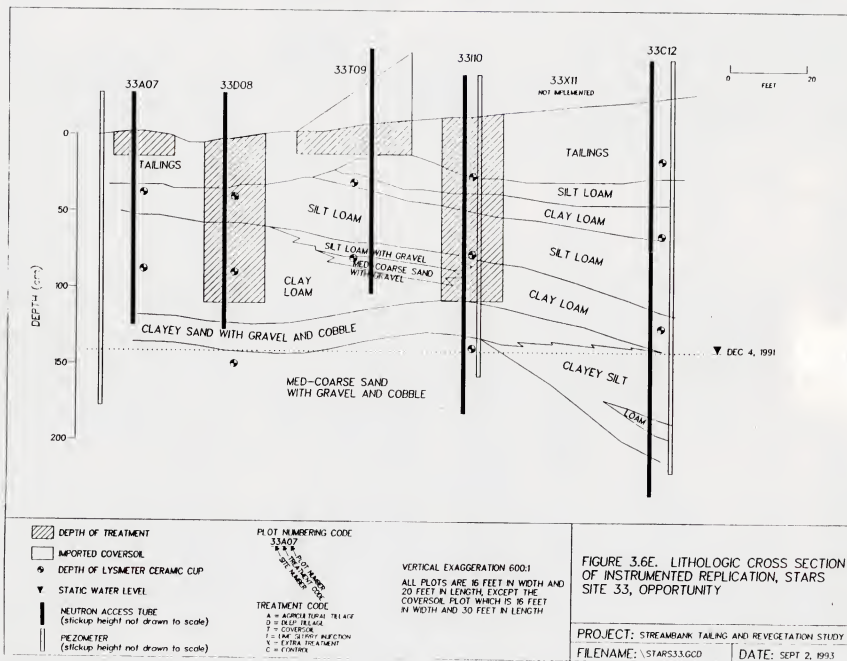


Figure 3.6e. Lithologic cross section of instrumented replication, STARS Site 33 (Opportunity).

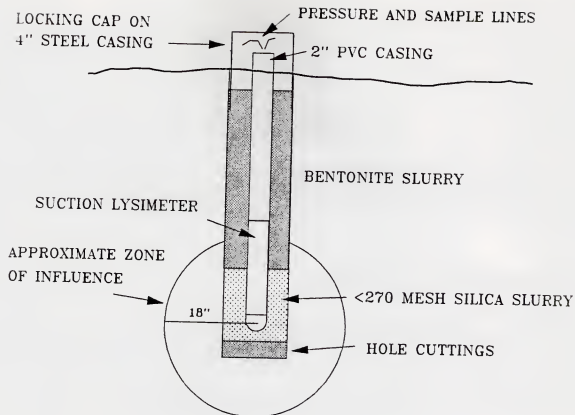


Figure 3.7. Schematic of pore water sampler installation.

Table 3.4. Soil chemistry monitoring.

Activity	Purpose	Frequency	Data Location
Soil pH/EC Monitoring	Assess tailings neutralization	1990 and 1991	B1
Water Soluble Metals	Assess metals movement to groundwater	1991	B1
Bio-Available Metals	Assess plant available metal levels	1991	B2
Surface Soil Metals	Assess before and after treatment surface metal levels	1992	B5
Amendment /Tailings Mixing	Photographic displays of degree of amendment mixing	1992	C4

The degree of mixing in the STARS plots was evaluated in the final year of the study (1992). Excavations were made to a depth of 0.6 to 2.1 meters (two to seven feet) in several STARS plots. A pH indicator dye was sprayed on the face of the excavation (approximately four feet wide by 5 feet deep) and a white silica flour was spread over the dye to enhance the resulting color. Zones with a pH of less than 4.0 were bright yellow, pH 4.5-5.5 (yellowish green), pH 5.5-7.0 (dark olive green), pH 7.0-8.0 (blue), and soils with a pH of greater than 8.0 were bluish-purple. This method allowed the project team to qualitatively describe the degree of mixing achieved by the incorporation methods, to describe the volume of soil neutralized, and to describe the continuity of acid and neutral zones. Photographs and schematic drawings were prepared from each excavation and are presented in Appendix C4.

3.5 VEGETATION MONITORING

The main objective for measuring vegetation performance was to determine the plant species' ability to germinate, survive, and reproduce in the altered environments prepared at the five STARS sites. To determine whether the remedial technology will promote a productive and persistent vegetation community, five major monitoring activities were implemented. Each is identified in Table 3.5 along with the purpose for generating the data, frequency of collection, and appendix location of the data.

Table 3.5. Vegetation monitoring.

Activity	Purpose	Frequency	Data Location
Plant Density	Assess vegetation establishment and production of viable seed	1989	D1
Areal Plant Cover	Assess reduction of contaminant movement	1989, 1990, and 1991	D2, D3, D4
Vegetation Production	Guide species selection for waste stabilization	1991	D5
Vegetation Metal/As Loadings	Assess potential for live-stock/wildlife bioaccumulation	1992	D7
Rooting Characteristics	Assess long-term community stability	1992	D6

Percent areal cover, measured in all three years, indicated the relative effectiveness of the establishing vegetation to reduce leaching, surface runoff, and fugitive dust. Plant production (peak standing crop) indicated potential future land uses of the tailings once they are successfully reclaimed, assuming plant metal levels are not toxic for grazing animals. Rooting depths indicated long-term plant community stability; plants with root systems limited to upper soil layers would be less drought tolerant than those that can penetrate deeper layers. When combined with soil metal levels, plant metal levels indicate whether plants are accumulating metals, possibly passing them into the food chain, or excluding them. If plants accumulate metals, there may be a redistribution of metals from subsurface to surface soil layers as plant materials decompose.

Seed germination and seedling survival will indicate the relative success of amendments to allow vegetation establishment the first year, and for the established vegetation to set and disperse viable seed at the end of the second growing season.

3.6 DATA VALIDATION

Data generated during the monitoring phase of this project were under the control of the STARS Phase III QAPP document (Reclamation Research Unit, Schafer & Associates, and CH2M Hill 1989b). Laboratory analytical requests were prepared which defined analytical methods, technical requirements, quality control audits, and reporting requirements. These requests were consistent with requirements found in the QAPP document and were specifically written so that Data Quality Objectives (DQOs) also delineated in the QAPP were met. These requests were prepared using the same format used in the EPA's Contract Laboratory Program (CLP) for Special Analytical Services.

Data validation entailed a review of the quality control (QC) data and the raw laboratory data to verify that the laboratory was operating within required control limits, analytical results were correctly transcribed from instrument read-outs, and which, if any natural samples were related to any out-of-control QC results. The objective of data validation was to identify any unreliable or invalid laboratory measurements and to identify them using a data qualifier system.

Upon receipt of STARS data from a laboratory the following scheme was initiated:

- perform a technical review of the submitted data using technical requirements stipulated in the Analytical Request.
- determine if any natural samples were associated with out-of-control analytical situations, and to identify these by means of a data qualifier (descriptor) system.

Data packages received were evaluated using specific guidelines (as much as possible) established by EPA in "Laboratory Data Validation: Functional Guidelines for evaluating

Inorganic Analyses, Revised Draft (EPA 1988b). Summary of data QC screening forms supplied by the CLP Deputy Project Officer for EPA Region VIII were used to record these activities. All data packages were reviewed using the Analytical Request, the Functional Guidelines, and the QC Summary forms. Major requirements evaluated during data review included: sample holding times, instrument calibrations, preparation blanks, laboratory control sample results, results of natural samples including field QC samples (replicates, standard reference materials, field blanks), laboratory QC sample results (spikes and duplicates), ICP interference check sample results, furnace atomic absorption QC results, ICP serial dilution results, and the overall assessment of data.

Field and laboratory QC sample results were then used to quantify the accuracy, precision, comparability, and completeness of the data. Representativeness of the data was qualitatively assessed. The results of the validation process was a series of DATA Validation Reports containing data tables showing analytical values and quality descriptors, a narrative describing the quality of the data, statements regarding the useability of the data in terms of the established DQOs, statistical calculations describing the accuracy, precision, completeness, and comparability of the data, and the EPA Region VIII summary forms. These Data Validation Reports are part of the project files, copies are available at the offices of the Montana Department of Health & Environmental Sciences.

3.7 PERFORMANCE APPRAISAL OF THE STARS REMEDIAL ALTERNATIVE

In the Feasibility Study report for the Streamside Tailings Operable Unit, a variety of remedial alternatives will be compared using the Superfund required evaluation criteria listed in Table 3.6. Data needed to compare the STARS remedial measures with other potential remedial options were collected as part of this Phase III STARS monitoring program. Performance data measured were selected based on data needs for completing the FS. The data needs of many evaluation criteria were already available. For example, the implementation and costs of the STARS remedial methods have already been described in the Phase II STARS Final Report (Schafer and Associates, Reclamation Research, and CH₂M HILL, 1989b). These estimates have been updated and are presented in this report in Section 4.4. State and community acceptance will be determined by public hearing and by administrative review of the FS. The Phase III monitoring program is designed to provide data to evaluate short-and long-term effectiveness; reduction in toxicity, and mobility; protection of public health and the environment; and attainment of clean-up goals (project-specific "ARARS" or applicable and relevant, or appropriate requirements).

The primary objective of the STARS treatments is to restrict the mobility of contaminants in tailings and to reduce their phytotoxicity. STARS treatments were designed so that mobility was decreased by two processes. First, mobility is decreased by reducing the solubility of metals in tailings. Reducing metal solubility should reduce concentration of metals in leachate, surface runoff, and vegetation. Second, mobility should be controlled by

reducing the magnitude of specific release mechanisms. For example, revegetating tailings is expected to:

- Reduce the quantity of eroded material by increasing flow resistance and by stabilizing the soil surface;
- Reduce leachate to groundwater by increasing plant water use;
- Reduce surface runoff by increasing soil infiltration rate; and
- Decrease fugitive dust.

The effectiveness of the STARS technology is evaluated in the following sections of this report. It is quantified by how much the mobility and availability of metals is reduced. Performance data generated in the monitoring program included measured or estimated changes in metal movement caused by surface runoff, erosion, leachate, fugitive dust, and bioaccumulation.

The short- and long-term effectiveness of STARS remedial measures were also evaluated. Short-term effectiveness was observed through the first two years of the study. Long-term effectiveness was estimated by applying appropriate models calibrated to the Silver Bow Creek site. Specific models appropriate for this site were selected. Input requirements for the model were assessed to be sure that the correct data were being collected.

Table 3.6. Criteria for evaluating and selecting remedial measures during the FS.

Threshold Criteria:

- Overall protection of human health and environment
- Compliance with ARARs

Balancing Criteria:

- Short-term effectiveness
- Long-term effectiveness
- Reduction in toxicity, mobility, and volume
- Implementability
- Cost

Modifying Criteria:

- State acceptance
 - Community acceptance
-

The STARS technology may reduce erosion and sedimentation during 10 or 100 year floods by stabilizing surface materials. During a 100 year flood event, erosion and sedimentation of sandy tailings materials in and immediately adjacent to active channels may occur. Areas above active channels will not likely be exposed to significant erosion due to the shallow depth and lower velocity of overbank flood flows. STARS quantitative vegetative cover data may provide input data to model expected flood-caused erosion in the FS, but this data alone is insufficient to address the long-term effectiveness of STARS technology in reducing flood-induced erosion and sedimentation. The long-term effectiveness of STARS amendments in such situations may be enhanced by amending the full depth of tailings materials in areas subject to high erosion potential (determined through flood flow/sedimentation modeling). This could leave an acceptable substrate in which a new vegetative cover could reestablish.

4.0 PHASE III MONITORING RESULTS

Section 4.0 presents the results of three years (1990 through 1992) of experimental monitoring. The results are reported in three major subsections including soil hydrology, which addresses soil water content (neutron probe data), depth to groundwater, climatic data, and simulated rainfall runoff test results; soil chemistry monitoring, which presents total, extractable and water soluble metals, pH and EC results, pore water chemistry as well as an evaluation of amendment mixing; and vegetation monitoring, which presents data on seed germination and survival, plant cover, production and rooting characteristics. The discussion of results focuses on comparisons of the relative effectiveness of each treatment at obtaining the objectives outlined in section 3.0. The role of each monitoring component as it applies to the conceptual model (section 2.0) is also discussed.

4.1 SOIL HYDROLOGICAL MONITORING

Soil hydrology plays an important role in contaminant fate and transport. Chemical interactions between soil solution and the soil/tailings matrix and the unsaturated flow of water in the vadose zone will ultimately determine the composition and transport of water migrating to groundwater. Similarly, climatic factors including precipitation intensity, duration and timing, as well as evapotranspiration will greatly influence the quantity, degree and quality of runoff waters from tailings. This subsection presents and discusses the results of the soil hydrological monitoring program.

4.1.1 Soil Water Content

Soil water content was measured on each field sampling date using neutron attenuation. Water monitoring began in December 1989 and continued until mid 1992. Equipment installation and calibration is described in more detail in section 3.1.

Data Interpretation: The purpose of measuring soil water content was to evaluate changes in soil water content as a means of investigating the site hydrologic balance. In addition, the pattern of spatial and temporal changes in water content provided a great deal of information about the soil-plant system.

Within the vadose zone (*eg.* above the capillary fringe) the measured water content at a particular depth is a function of the grain size and the matric suction. Soils that have not been affected by a recent recharge event or by evaporative drying from the soil surface or by plant water uptake normally have a suction of 0.1 to 0.3 bars. The moisture content at this level of suction is termed the "field capacity". Under conditions of uniform suction, water content is purely a reflection of grain size. Coarse-textured soils usually have a volumetric water content of 20 percent or less while finer-textured soils traditionally have

water content of 25 to 35 percent under these conditions. Variations in water content with depth which remain unchanged between dates of measurement are a reflection of textural stratification.

Temporal changes in water content at a specific depth are indicative of the hydrologic process causing the change in water content. Progressive drying of surface layers with diminishing differences at depth indicate soil surface evaporation (to a depth of 15 cm) or plant water use (Figure 4.1.1) (to 150 cm in depth or more). The lower limit of water content reached when plants have extracted all "available" water is termed the "wilting point", and the net difference between field capacity and the wilting point is the water-holding capacity. Replenishment of soil water can be observed as a sequential downward migration of a discrete "wetting front" (Figure 4.1.2). The effects of a water table and capillary fringe can also be observed using neutron probe data. Depth to groundwater is about 180 cm as shown in Figures 4.1.1 and 4.1.2. Due to fine-textured particle size of soils at this site, capillary rise of water caused nearly-saturated conditions to exist well above the water table (between 115 and 180 cm).

Manganese Stockpile: Vegetation performance was poor on all plots at the Manganese Stockpile except the cover-soil treatment. As a result, hydrologic monitoring was discontinued after February, 1991. All plots (Appendix A-2, Neutron Probe Results) showed the same general trends in water content. Little or no change in water content was observed between dates suggesting that the tailings were near field capacity throughout the year. The water table at the Manganese Stockpile was closest to the soil surface of all sites monitored. As a consequence, the water potential varied from field capacity at a depth of 1 to 2 feet to near saturation at the groundwater level. Due to high water content at the Manganese Stockpile site, small amounts of infiltrating water may have recharged the groundwater system. High water content also would tend to increase the runoff rate during precipitation events. The poor drainage and periodic flooding at the site may have contributed to failure of the stand on most plots.

The stratification of the tailings and natural soil was also evident with a layer of coarse tailings (0 to 45 cm) over a layer of nearly-saturated slime tailings (45 to 90 cm). Another thin layer of sandy tailings was found from 90 to 120 cm which in turn was underlain by sandy natural soil which was generally saturated most of the year. Deep tillage (plot 24) was effective in removing the stratification from the upper 100 cm of the soil profile due to the tillage-induced mixing of contrasting soil horizons. Such mixing would have the effect of increasing the rate of internal drainage in deep-plowed soils.

Ramsay Flats, Plot 17, Deep Plow

VOLUMETRIC WATER CONTENT (CM3/CM3)

DEPTH (cm)

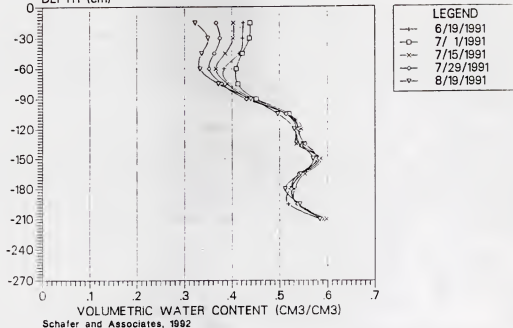


Figure 4.1.1.

Progressive decreases in water content in the upper 90 cm of the deep plow plot at Ramsay shows the effect of plant water extraction.

Ramsay Flats, Plot 17, Deep Plow

VOLUMETRIC WATER CONTENT (CM3/CM3)

DEPTH (cm)

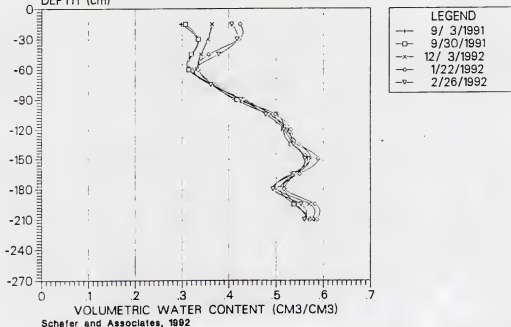


Figure 4.1.2.

A modest amount of recharge of the soil profile dried by evapotranspiration is recharged between 12/91 and 2/92. The dry soil "reservoir" prevented groundwater recharge.

Rocker: The Rocker site consisted of an average of 120 cm of gravelly coarse sand over a layer of silty natural soil. "Tailings" at the Rocker site included mixed mine waste and alluvium to a depth of 50 to 100 cm or more. It was difficult to visually distinguish between natural alluvium and contaminated material at this site although surficial tailings were strongly iron-stained. Buried natural soils consisted of thinly stratified medium sands and silty fine sands.

Soil water content increased from less than 10 to 15 percent near the soil surface to over 50 percent at depth (Appendix A-2). The substantial increase noted in all plots at 90 to 110 cm was due to capillary rise from the water table [1] (Hillel 1980) measured seasonally at 125 to 150 cm below ground surface (Figure 4.1.3). This magnitude of capillary rise (35 to 40 cm) equated to an average pore diameter of 0.05 mm or very fine sand sized.

$$H_c = \frac{2\sigma \cos \alpha}{r \rho_w g}$$

where σ = surface tension

α = wetting angle

r = pore radius

ρ_w = density of water

[1]

$$H_c (cm) = \frac{0.153}{r} (cm)$$

Soils were somewhat stratified at Rocker as at the Manganese Stockpile site except for the deep plow plot which had uniform characteristics within the zone of tillage (100 cm). The water table at Rocker as elsewhere along Silver Bow Creek probably responded to stream stage in that groundwater dropped by 30 to 40 cm between its highest point in June to its lowest in October (Appendix A-2).

Water content changed seasonally at Rocker in response to evaporation, and transpiration withdrawal of soil water from June through September. Gradual recharge of the soil profile occurred from September through the following April (Figure 4.1.4). In order to develop the information presented in Figure 4.1.4, the total amount of water in each soil profile was summed for the depth to which neutron probe measurements were taken. Differences in stored soil water reflected the seasonal flux of water within each of the test plots. The net change in water content observed in each plot was a function of the soils water-holding capacity and the depth to which evaporation and transpiration reached. Due to the low water-holding capacity of the coarser soils found at Rocker compared to other STARS sites, the seasonal change in water content was less at Rocker than elsewhere.

Rocker - Depth to Groundwater

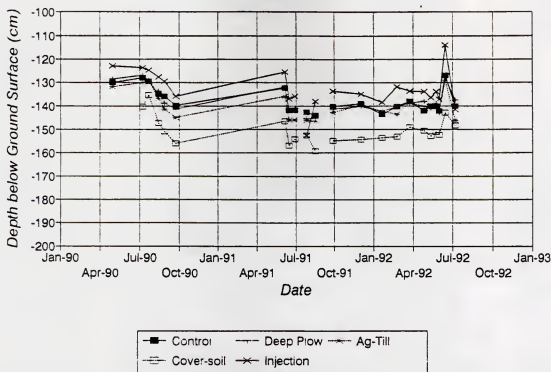


Figure 4.1.3. Seasonal fluctuation in the depth to groundwater measured at Rocker.

The control plot exhibited the least variation in seasonal water content while the cover-soil and deep tillage plots, those with the best vegetative cover, had the widest variation in water content. The shallow groundwater at the site probably contributed to the success of vegetation on plots where it became established.

As a consequence of the limited water-holding capacity of soils at Rocker and due to the more rapid infiltration rate, more water appeared to percolate through Rocker soils than at the other STARS sites (Table 4.1.1 and Table 4.1.2). A water balance for the unvegetated control indicates that while much of the 33.8 inches of rainfall recorded over a two-year period evaporated from the site (24.9 inches), only 1.0 inches of water was predicted to run off the site and the remaining 6.5 inches percolated through the soil and recharged the shallow groundwater system. Recharge was thought to be a very localized short-duration event which occurred sporadically in late winter through early summer. A similar water balance developed for the grass-covered deep plow plot indicated that a smaller quantity of percolation occurred (4.7 inches over 2 years) and a proportionately larger fraction of soil water was consumed by evapotranspiration. Establishment of a vegetative cover reduced the net percolation to groundwater at this site as expected. Established vegetation was able to withdraw water from the upper 18 to 24 inches of the root zone during the growing season at the Rocker site (Appendix A-2). Water use from deeper in the profile may also have occurred but without a net change in the water content due to upward flux from groundwater into the capillary fringe. This process is common in sub-irrigated areas.

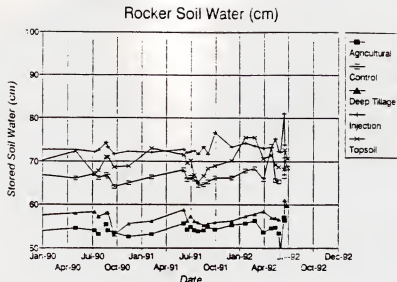


Figure 4.1.4. Comparison of seasonal changes in stored soil water within the soil root zone in each of the treatments at the STARS Rocker site.

The water balance for the control and deep plow plots presented for the Rocker, Ramsay, and Opportunity sites was prepared by first determining the net change in water content in neutron probe tubes between monitoring dates. Incremental rainfall for the same period was based on the storage or recording rain gauge at the records at each site. Where rain gauge data were incomplete, precipitation data were used from the nearest station with record. Runoff was predicted using GLEAMS (see section 5.1) for each daily rainfall event. Daily rainfall amounts were taken from the Ramsay Flats precipitation records. For each time increment between monitoring dates, the change in soil water content and the calculated runoff was subtracted from total precipitation. The quantity of "excess precipitation" was ascribed to either evaporation or percolation depending on the value for potential evaporation. When potential evaporation was equal to or greater than the excess precipitation, then evaporation was assumed to account for the consumption of excess water. When the difference was greater than potential evaporation then percolation was assumed to occur. Cumulative errors of up to several inches of percolation are inherent in this water balance method.

Ramsay Flats: A large volume of tailings were deposited in a low gradient reach of Silver Bow Creek in the Ramsay Flats area. Ramsay Flats tailings extend several hundred feet north of the Silver Bow channel. About 50 cm of silty tailings overlie texturally stratified natural sediments consisting of low permeability silts and silty-clays, muck layers, and stringers of fine sand. The potentiometric surface (Figure 4.1.5) varied from 180 to 210 cm below the soil surface, and varied seasonally and assumed to be in response to changes in the stage of Silver Bow Creek. Hydraulic connection between Silver Bow Creek and shallow groundwater below Ramsay Flats was inferred due to the similarity in the seasonal trend of piezometer readings to Opportunity and Rocker where hydraulic connection with the creek was clear.

Table 4.1.1. Hydrologic water balance for the Rocker control plot.

DATE	PERIOD RAINFALL	SOIL WATER CHANGE	ESTIMATED RUNOFF	NET DIFFERENCE	PET (in)	ESTIMATED ET	ESTIMATED PERC
Measured in inches							
02/22/90	1.34						
05/02/90	0.75	-0.28	0.44	0.58	0.58	0.58	0.03
07/11/90	3.78	0.39	0.00	3.39	8.11	3.39	0.00
07/25/90	0.63	-0.35	0.00	0.98	2.71	0.98	0.00
08/15/90	2.91	0.00	0.00	2.91	5.40	2.91	0.00
08/23/90	1.18	0.28	0.00	0.91		0.00	0.91
08/26/90	1.08	-0.18	0.00	1.22	1.38	1.22	0.00
09/24/90	0.11	-0.91	0.00	1.02	3.80	1.02	0.00
10/24/90	0.41	0.31	0.00	0.09	1.00	0.09	0.00
02/05/91	1.28	0.55	0.00	0.71	0.33	0.33	0.38
06/07/91	5.98	0.63	0.33	5.02	3.58	3.58	1.45
06/17/91	0.12	-0.87	0.00	0.96	1.54	0.96	0.00
07/01/91	2.24	0.20	0.00	2.05	1.08	1.08	0.97
07/15/91	0.35	-0.18	0.00	0.51	2.98	0.51	0.00
07/29/91	0.28	-0.55	0.00	0.83	2.80	0.63	0.00
08/19/91	0.12	0.08	0.00	0.04	3.95	0.04	0.00
09/03/91	0.79	0.24	0.00	0.55	2.45	0.55	0.00
09/30/91	1.22	0.35	0.00	0.87	2.04	0.87	0.00
12/04/91	1.73	0.00	0.00	1.73	1.05	1.05	0.68
01/23/92	0.39	0.63	0.00	-0.24	0.01	-0.24	0.00
02/26/92	0.35	0.24	0.00	0.12	0.14	0.12	0.00
03/26/92	0.51	-0.16	0.00	0.67	0.47	0.47	0.20
04/28/92	0.39	2.20	0.20	-2.01	1.32	-2.01	0.00
05/14/92	0.08	-3.11	0.00	3.19	1.58	1.58	1.63
05/28/92	0.43	-0.12	0.00	0.55	1.54	0.55	0.00
06/02/92	0.39	0.00	0.00	0.39	0.90	0.39	0.00
06/16/92	2.52	0.47	0.00	2.05	1.79	1.79	0.26
07/08/92	3.82	1.54	0.00	2.28	8.11	2.28	0.00
	33.8	1.8	1.0	31.4	60.5	24.9	6.5

Table 4.1.2. Hydrologic water balance for the Rocker deep plow plot.

DATE	PERIOD RAINFALL	SOIL WATER CHANGE	ESTIMATED RUNOFF	NET DIFFERENCE	PET (in)	ESTIMATED ET	ESTIMATED PERC
Measured in inches							
02/22/90	1.34						
05/02/90	0.75	0.20	0.51	0.04	0.58	0.04	0.00
07/11/90	3.78	0.12	0.00	3.86	8.11	3.86	0.00
07/25/90	0.63	-0.43	0.00	1.06	2.71	1.06	0.00
08/15/90	2.91	0.18	0.00	2.78	5.40	2.78	0.00
08/23/90	1.18	0.16	0.00	1.02		0.00	1.02
08/26/90	1.08	0.04	0.00	1.02	1.38	1.02	0.00
09/24/90	0.11	-2.01	0.00	2.12	3.80	2.12	0.00
10/24/90	0.41	1.02	0.00	-0.62	1.00	-0.62	0.00
02/05/91	1.28	0.24	0.00	1.02	0.33	0.33	0.69
06/07/91	5.98	1.02	0.37	4.59	3.58	3.58	1.01
06/17/91	0.12	-0.98	0.00	1.10	1.54	1.10	0.00
07/01/91	2.24	0.39	0.00	1.85	1.08	1.08	0.77
07/15/91	0.35	-0.43	0.00	0.79	2.98	0.79	0.00
07/29/91	0.28	-0.12	0.00	0.39	2.80	0.39	0.00
08/19/91	0.12	-0.24	0.00	0.35	3.95	0.35	0.00
09/03/91	0.79	0.12	0.00	0.67	2.45	0.67	0.00
09/30/91	1.22	0.12	0.00	1.10	2.04	1.10	0.00
12/04/91	1.73	0.12	0.00	1.81	1.05	1.05	0.58
01/23/92	0.39	0.43	0.00	-0.04	0.01	-0.04	0.00
02/26/92	0.35	0.20	0.00	0.18	0.14	0.14	0.02
03/26/92	0.51	0.24	0.00	0.28	0.47	0.28	0.00
04/28/92	0.39	-0.55	0.15	0.79	1.32	0.79	0.00
05/14/92	0.08	-0.94	0.00	0.12	1.58	0.12	0.00
05/28/92	0.43	-0.12	0.00	0.55	1.54	0.55	0.00
06/02/92	0.39	0.08	0.00	0.31	0.90	0.31	0.00
06/16/92	2.52	0.12	0.00	2.40	1.79	1.79	0.81
07/08/92	3.82	1.57	0.00	2.24	8.11	2.24	0.00
	33.8	1.4	1.0	31.4	60.5	26.7	4.7

Capillary rise in the fine-grained sediments (estimated mean pore size of 5 to 10 microns) was predicted to be as much as 1.5 to 3 m, hence the water table influence could have reached to the soil surface at Ramsay. Gradual upward movement of water from the water table due to capillary rise may account for the formation of a salt crust at the soil surface in summer at Ramsay and in other locations along Silver Bow Creek and the upper Clark Fork River.

Capillary rise may also account for the uniform water content approaching 50 % in the injection and ag-till plots (Appendix A-2). Coarser-textured stratifications are evident in the control and cover-soil plots at 50 and 100 cm respectively. The deep plow plot had a much lower average water content in the upper 100 cm in response to tillage which reduced the soil bulk density hence increasing the average pore size.

The total quantity of stored soil water in Ramsay Flats plots reflected the effectiveness of vegetation in removing water from the soil profile during the growing season (Figure 4.1.6). Water was removed to a depth of nearly three feet in plots with the best vegetation. The resulting water deficit was subsequently recharged the following winter and spring. Very little net change in water content occurred in the control plot due to the limited vertical influence of evaporation from dry soil (Hillel 1980). Plots with healthy vegetation exhibited a larger soil water deficit, especially in 1991 which had a dry summer. The rate at which water was withdrawn from the soil appeared to be the greatest in those plots with the best vegetative cover (eg. cover-soil, deep plow, and ag-till).

A water balance was developed for the control and deep plow plots at Ramsay (Table 4.1.3 and 4.1.4). The net percolation or recharge to groundwater (corrected for neutron probe reading errors in August, 1990 and January, 1992) was about 1.4 inches for the control plot with a similar estimate from the deep-plow plot. This amount of water estimated over a two-year period (0.7 inches per year) is fairly insignificant and may be within the range of cumulative error inherent in this method for estimating percolation. The smaller amount of percolation through the soils at Ramsay than that observed at Rocker was expected due to the low permeability of the materials and to the capillary influence of the groundwater system which may have resulted in a very slow net upward flux of water into the soil profile for much of the year.

Ramsay Flats - Depth to Groundwater

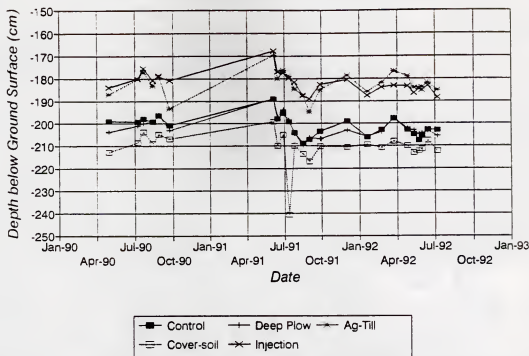


Figure 4.1.5.

Seasonal fluctuation in the depth to groundwater measured at Ramsay Flats.

Ramsay Flats Soil Water (cm)

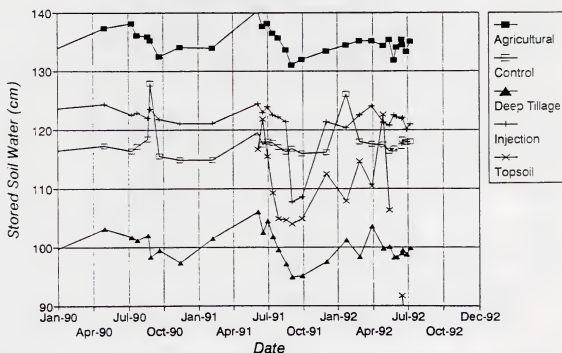


Figure 4.1.6.

Comparison of seasonal changes in stored soil water within the soil root zone in each of the treatments at the STARS Ramsay Flats site.

Table 4.1.3. Hydrologic water balance for the Ramsay Flats control plot.

DATE	PERIOD RAINFALL	SOIL WATER CHANGE	ESTIMATED RUNOFF	NET DIFFERENCE	PET (in)	ESTIMATED ET	ESTIMATED PERC
Measured in inches							
05/02/90	1.17	0.4	0.09	0.88	0.56	0.56	0.13
07/11/90	2.88	-0.4	0	3.28	8.11	3.28	0.00
07/25/90	0.35	0.4	0	0.00	2.71	0.00	0.00
08/23/90	0.00	0.5	0	-0.47	5.40	-0.47	0.00
08/28/90	0.00	-0.2	0	0.20	1.36	0.20	0.00
09/24/90	0.01	-0.9	0	0.95	3.80	0.95	0.00
10/24/90	0.00	-0.3	0	0.28	1.00	0.28	0.00
02/06/91	0.57	0.0	0	0.57	0.33	0.33	0.24
06/07/91	5.82	1.9	0.64	3.13	3.58	3.13	0.00
06/17/91	0.10	-0.8	0	0.89	1.54	0.89	0.00
07/01/91	1.65	0.2	0	1.45	1.08	1.08	0.37
07/15/91	0.19	-0.1	0	0.27	2.96	0.27	0.00
07/29/91	0.19	-0.3	0	0.50	2.80	0.50	0.00
08/19/91	0.48	-0.3	0	0.78	3.95	0.78	0.00
09/03/91	0.56	0.2	0	0.39	2.45	0.39	0.00
09/30/91	1.57	-0.3	0.05	1.84	2.04	1.84	0.00
12/04/91	0.73	0.1	0	0.85	1.05	0.85	0.00
01/23/92	0.87	0.1	0	0.76	0.01	0.01	0.75
02/26/92	0.29	0.8	0	-0.34	0.14	-0.34	0.00
03/26/92	0.71	-0.2	0.03	0.84	0.47	0.47	0.37
04/28/92	0.19	-0.0	0.59	-0.36	1.32	-0.36	0.00
05/14/92	0.06	-0.5	0	0.53	1.58	0.53	0.00
05/26/92	0.35	0.2	0	0.19	1.54	0.19	0.00
06/02/92	0.42	0	0	0.42	0.90	0.42	0.00
06/16/92	1.89	0.2	0.2	1.54	1.79	1.54	0.00
	20.88	0.32	1.80	18.95	52.44	17.10	1.85

Table 4.1.4. Hydrologic water balance for the Ramsay Flats deep plow plot.

DATE	PERIOD RAINFALL	SOIL WATER CHANGE	ESTIMATED RUNOFF	NET DIFFERENCE	PET (in)	ESTIMATED ET	ESTIMATED PERC
Measured in inches							
05/02/90	1.17	1.5	0.03	-0.36	0.56	-0.36	0.00
07/11/90	2.88	-0.8	0	3.43	8.11	3.43	0.00
07/25/90	0.35	-0.2	0	0.55	2.71	0.55	0.00
08/23/90	0.00	0.3	0	-0.31	5.40	-0.31	0.00
08/28/90	0.00	-1.5	0	1.46	1.38	1.38	0.10
09/24/90	0.01	0.5	0	-0.46	3.80	-0.46	0.00
10/24/90	0.00	-0.9	0	0.87	1.00	0.87	0.00
02/06/91	0.57	1.7	0	-1.08	0.33	-1.08	0.00
06/07/91	5.82	1.8	0.2	3.65	3.58	3.58	0.07
06/17/91	0.10	-1.4	0	1.48	1.54	1.48	0.00
07/01/91	1.65	0.8	0	0.86	1.08	0.88	0.00
07/15/91	0.19	-1.1	0	1.28	2.98	1.28	0.00
07/29/91	0.19	-0.9	0	1.06	2.80	1.06	0.00
08/19/91	0.48	-1.0	0	1.45	3.95	1.45	0.00
09/03/91	0.58	-0.9	0	1.45	2.45	1.45	0.00
09/30/91	1.57	0.1	0	1.49	2.04	1.49	0.00
12/04/91	0.73	0.8	0	-0.21	1.05	-0.21	0.00
01/23/92	0.87	1.5	0	-0.58	0.01	-0.58	0.00
02/26/92	0.29	-1.1	0	1.43	0.14	0.14	1.29
03/26/92	0.71	2.0	0.08	-1.42	0.47	-1.42	0.00
04/28/92	0.19	-1.5	0.25	1.40	1.32	1.32	0.08
05/14/92	0.06	0.1	0	-0.06	1.58	-0.06	0.00
05/26/92	0.35	-0.7	0	1.10	1.54	1.10	0.00
06/02/92	0.42	0.0	0	0.42	0.90	0.42	0.00
06/16/92	1.89	0.3	0	1.82	1.79	1.62	0.00
	20.88	-0.15	0.56	20.47	52.44	18.93	1.24

Fairmont: The Fairmont site consisted of a thin and somewhat discontinuous deposit of tailings over historically flood-irrigated agricultural soils. The water table was quite deep (more than several meters) at Fairmont because the site is in an upland setting and is several thousand meters from the Silver Bow Creek floodplain. As a consequence, no capillary effects were observed in the Fairmont plots.

Soils at Fairmont consisted of a thin layer (20 to 60 cm) of loamy soil over alluvial terrace sand and gravel. The deep plow treatment at Fairmont extended to a more modest depth than at other sites due to the pervasive gravels.

Changes in stored soil water summed for the entire soil profile in each plot are depicted in Figure 4.1.7. The amount of soil water deficit which formed during the growing season, and the rate of water withdrawal correlated with the abundance of vegetation established on the plots.

Opportunity: Soils at the Opportunity site had a surficial layer of sandy tailings ranging from 20 to 40 cm thick. Natural soils were loamy in texture and had increasing amounts of gravels with depth. The average depth to groundwater was 140 to 190 cm at Opportunity (Figure 4.1.8). The potentiometric surface varied seasonally in response to changes in the stage of Silver Bow Creek. The apparent extent of capillary rise was nearly 100 cm suggesting an average pore diameter of 15 microns. The proximity of the water table to the soil surface indicated that sub-irrigation was likely to occur once vegetation became established.

Stored soil water content changes (Figure 4.1.9) at Opportunity followed the same pattern as those observed at other sites. Plots with an abundance of established vegetation withdrew more water more quickly during the summer than the control plot. A water balance developed for the control (Table 4.1.5) and deep plow plot (Table 4.1.6) indicated that approximately 3 inches of percolation may have occurred in both the control and in the deep plow plot.

Water Content Summary: Unvegetated tailings at all of the sites had measured water content at or above field capacity. The capillary fringe above the groundwater was detected at all streambank tailings sites except Fairmont which is in an upland setting. The high water contents which do not significantly vary seasonally indicate that bare tailings have little capacity to store infiltrating rainfall or snowmelt, hence increasing the probability of groundwater recharge. Water balance results, although somewhat inexact, suggest that 1.5 to 3 inches of recharge occurred annually in bare tailings. Neutron probe monitoring results as well as physical differences between the sites indicate that coarse tailings material represented by the Rocker site have the greatest potential for recharge while finer tailings as typified by the Ramsay site have the least risk. Evaporation from the bare soil surface only dried the soil profile of untreated tailings to several inches in depth. This finding is consistent with most references (Hillel 1980). In addition, unsaturated flow modeling of streambank tailings (Section 5.2) also confirmed that approximately 3 inches of groundwater recharge would occur annually in nonvegetated control plots.

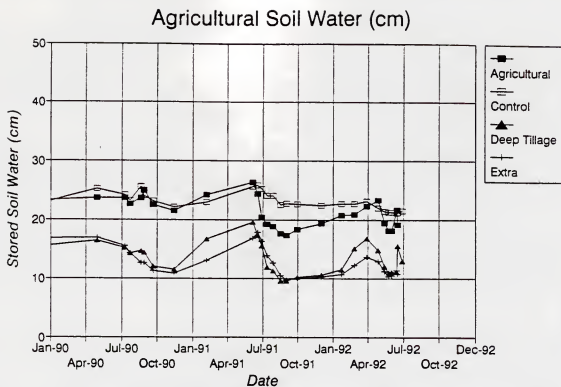


Figure 4.1.7. Comparison of seasonal changes in stored soil water within the soil root zone in each of the treatments at the STARS Fairmont site.

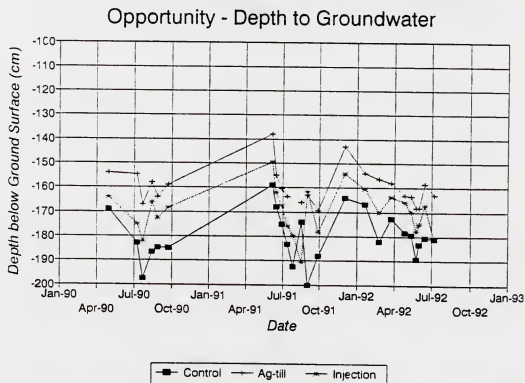


Figure 4.1.8. Seasonal fluctuation in the depth to groundwater measured at Opportunity.

Establishment of vegetation appeared to decrease the potential for recharge to occur. The magnitude in the reduction in recharge may be directly correlated with the productivity and vigor of the established vegetation. A correlation between the amount of water removed during the 1991 growing season and aboveground biomass production is shown in Figure 4.1.10. All control plots had low biomass production as well as small soil water deficits at the end of the 1991 growing season. Measured soil water deficit of 7 to 8 cm (3 inches, enough to prevent recharge) corresponded to plots with production of roughly 2200 to 2400 kg/ha of aboveground biomass. The increased plant water use on some plots may have been partially offset by an increase in infiltration rate, especially at Opportunity.

Opportunity Soil Water (cm)

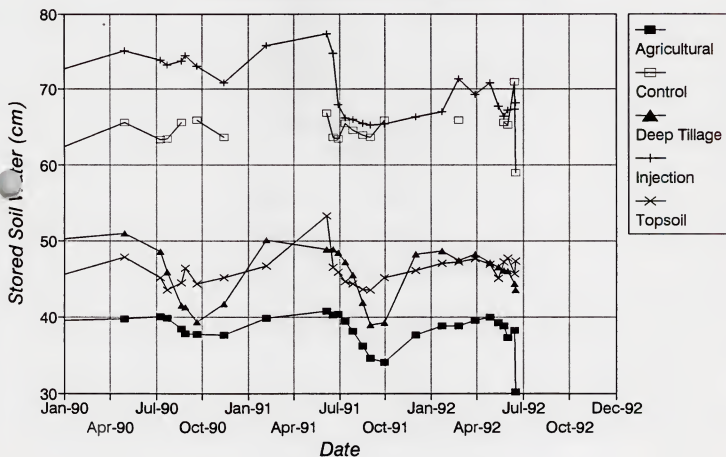


Figure 4.1.9.

Comparison of seasonal changes in stored soil water within the soil root zone in each of the treatments at the STARS Opportunity site.

Table 4.1.5. Hydrologic water balance for the Opportunity control plot.

DATE	PERIOD RAINFALL	SOIL WATER CHANGE	ESTIMATED RUNOFF	NET DIFFERENCE	PET (in)	ESTIMATED ET	ESTIMATED PERC
Measured in inches							
05/02/90	1.17	1.4	1.13	-1.33	0.56	-1.33	0.00
07/11/90	3.35	-0.9	0.4	3.61	8.11	3.81	0.00
07/25/90	0.26	0.0	0	0.24	2.71	0.24	0.00
08/23/90	0.67	0.8	0.04	0.00	5.40	0.00	0.00
08/28/90	0.67	0.0	0.01	0.62	1.36	0.62	0.00
09/24/90	0.37	0.1	0	0.29	3.80	0.29	0.00
10/24/90	0.26	-0.9	0	1.12	1.00	1.00	0.12
02/06/91	1.18	0.1	0	1.08	0.33	0.33	0.73
06/07/91	8.42	1.1	0.94	4.34	3.58	3.58	0.78
06/17/91	0.35	-1.3	0	1.81	1.54	1.54	0.07
07/01/91	0.75	-0.1	0.16	0.67	1.08	0.67	0.00
07/15/91	0.35	0.6	0	-0.43	2.98	-0.43	0.00
07/29/91	0.63	-0.4	0	0.98	2.60	0.98	0.00
08/19/91	0.43	-0.2	0	0.67	3.95	0.67	0.00
09/03/91	0.94	-0.1	0	1.06	2.45	1.06	0.00
09/30/91	1.22	0.9	0.06	0.26	2.04	0.26	0.00
12/04/91	1.46	-0.1	0.2	1.34	1.05	1.05	0.28
01/23/92	0.67	-0.1	0	0.75	0.01	0.01	0.74
02/26/92	0.26	0.2	0	0.12	0.14	0.12	0.00
03/26/92	0.91	0.2	0	0.75	0.47	0.47	0.28
04/28/92	0.67	0.1	0	0.59	1.32	0.59	0.00
05/14/92	0.04	-0.2	0	0.26	1.58	0.26	0.00
05/26/92	0.06	-0.1	0	0.20	1.54	0.20	0.00
06/02/92	0.42	-0.1	0	0.53	0.90	0.53	0.00
06/16/92	1.89	0.3	0	1.62	1.79	1.62	0.00
	25.85	1.54	2.94	21.37	52.44	18.38	2.99

Table 4.1.6. Hydrologic water balance for the Opportunity deep plow plot.

DATE	PERIOD RAINFALL	SOIL WATER CHANGE	ESTIMATED RUNOFF	NET DIFFERENCE	PET (in)	ESTIMATED ET	ESTIMATED PERC
Measured in inches							
05/02/90	1.17	0.0	0.44	0.73	0.56	0.56	0.16
07/11/90	3.35	0.6	0	2.55	6.11	2.55	0.00
07/25/90	0.26	-0.9	0	1.22	2.71	1.22	0.00
08/23/90	0.67	-1.1	0	1.93	5.40	1.93	0.00
08/28/90	0.67	-1.7	0	2.60	1.36	1.36	1.24
09/24/90	0.37	-0.1	0	0.45	3.80	0.45	0.00
10/24/90	0.26	-0.7	0	1.01	1.00	1.00	0.01
02/06/91	1.18	0.9	0	0.28	0.33	0.28	0.00
06/07/91	8.42	3.3	0.33	2.76	3.58	2.76	0.00
06/17/91	0.35	-0.5	0	0.83	1.54	0.83	0.00
07/01/91	0.75	0.0	0	0.75	1.08	0.75	0.00
07/15/91	0.35	-0.2	0	0.51	2.98	0.51	0.00
07/29/91	0.63	-0.5	0	1.14	2.60	1.14	0.00
08/19/91	0.43	-0.7	0	1.10	3.95	1.10	0.00
09/03/91	0.94	-1.4	0	2.36	2.45	2.36	0.00
09/30/91	1.22	-1.2	0	2.40	2.04	2.04	0.37
12/04/91	1.46	0.2	0	1.30	1.05	1.05	0.25
01/23/92	0.67	3.5	0	-2.67	0.01	-2.67	0.00
02/26/92	0.26	0.2	0	0.12	0.14	0.12	0.00
03/26/92	0.91	-0.5	0	1.42	0.47	0.47	0.95
04/28/92	0.67	0.4	0	0.31	1.32	0.31	0.00
05/14/92	0.04	-0.5	0	0.51	1.58	0.51	0.00
05/26/92	0.06	-0.2	0	0.31	1.54	0.31	0.00
06/02/92	0.42	-0.2	0	0.57	0.90	0.57	0.00
06/16/92	1.89	-0.0	0	1.93	1.79	1.79	0.14
	25.85	-1.17	0.77	26.24	52.44	23.11	3.13

Water Use vs Vegetation Production

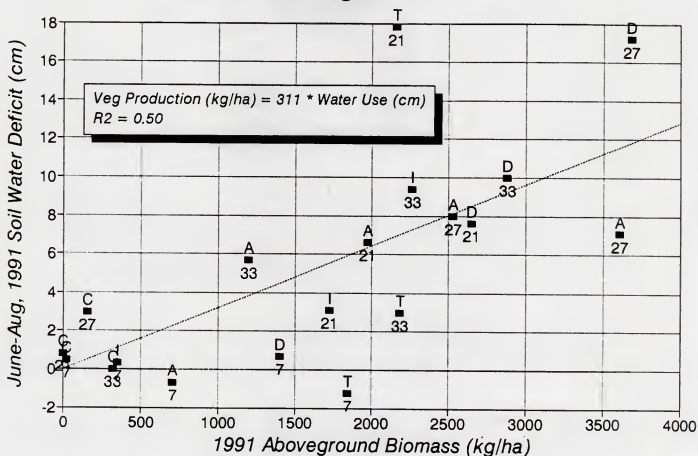


Figure 4.1.10.

Correlation between plant water use and biomass production for the 1991 growing season.

4.1.2 Climate Monitoring

Climatological Monitoring Network: Climatological monitoring was conducted at each of the STARS sites in order to evaluate the hydrologic water balance. Both precipitation, and evaporation are climatic variables of importance in determining the site water balance and although evaporation cannot be measured directly, the potential evapotranspiration can be estimated by measuring solar radiation and daily temperature.

A continuous climatic station was installed at Ramsay Flats which was in the central reach of Silver Bow Creek being investigated. At other sites, a storage rain gauge was installed. The climate station was equipped with sensors and a data-logger to measure maximum, minimum and average daily temperature, rainfall, daily solar radiation, and average daily wind speed. Rainfall and snowfall intensity data was collected using a propane-heated tipping-bucket gauge. When a precipitation event began, a time clock was triggered to measure cumulative precipitation every 5 minutes throughout the storm duration.

The Ramsay Flats station was installed on October 12, 1989 and operated up through early July, 1992. The station was down from July 24, 1991 through October 28, 1991 due to vandalism of the solar panel used to recharge the system battery. A small hiatus also occurred from November 14 through 20, 1991 when the power supply was refurbished. The rain gauge failed at the end of June, 1992 but temperature and radiation data were collected until early August, 1992.

Data from two NOAA weather stations, one in Butte and one in Anaconda were also acquired for use in hydrologic modeling at the STARS site. Daily climate data from both stations were acquired in magnetic format for the period of record from 1989 through mid 1992.

Climatic Characterization: Daily rainfall at Ramsay Flats is shown in Figure 4.1.11. Detailed climatological records are provided in Appendix A-1. Peak seasonal rainfall occurred in April through June as is typical for the intermountain region. Peak daily rainfall quantities ranged from 0.5 to 1.0 inches per day with a total of 6 events of more than 0.5 inches per day during 30 months of record. With one exception (an October snowfall), the peak rainfall intensity was measured from June through September (eg. convective thunderstorms). High intensity thunderstorms have historically been associated with maximum loading of metals to surface water resulting in historic fish kills on the upper Clark Fork. Rainfall hydrographs for three of the largest rainfall events are provided in Appendix A-1. Peak 5-minute rainfall intensity for the period of record was 2 to 3 inches per hour. Peak recorded 30-minute rainfall intensity was 1.2 inches per hour on 6/28/92.

Monthly precipitation was compared at each of the NOAA sites, the storage rain gauge locations, and at Ramsay Flats (Table 4.1.7). Differences in rainfall conformed to the spatial differences in average annual rainfall shown for the upper Clark Fork basin (SCS,

Rainfall Comparison

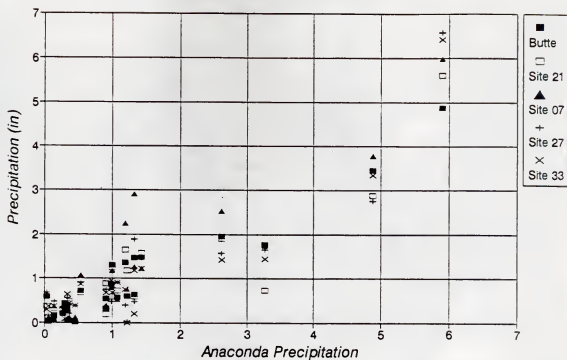


Figure 4.1.12.

Correlation of monthly precipitation at Anaconda with other weather stations.

RAMSAY FLATS CLIMATE STATION

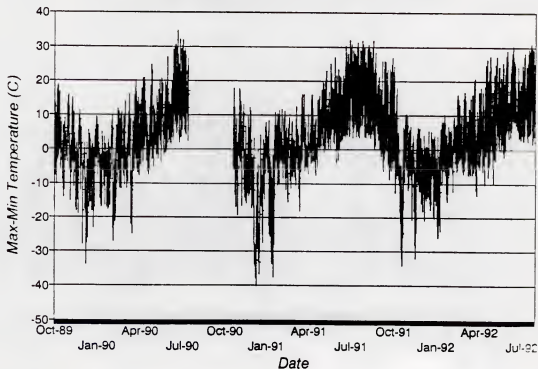


Figure 4.1.13.

Maximum and minimum daily temperatures at Ramsay Flats.

RAMSAY FLATS CLIMATE STATION

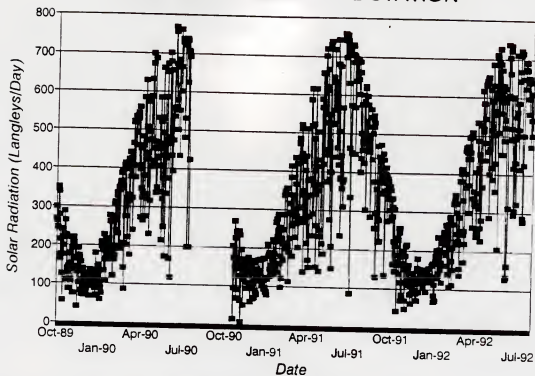


Figure 4.1.14. Daily solar radiation at Ramsay Flats.

RAMSAY FLATS CLIMATE STATION

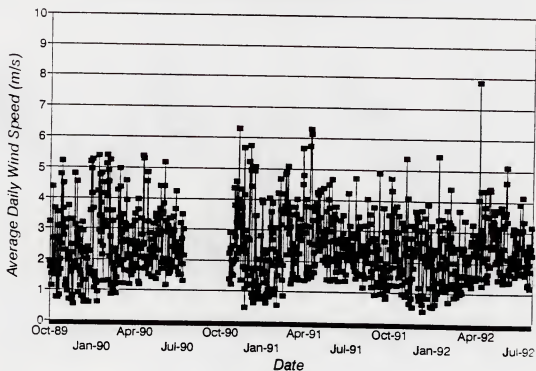


Figure 4.1.15. Average daily wind speed at Ramsay Flats.

RAMSAY FLATS CLIMATE STATION

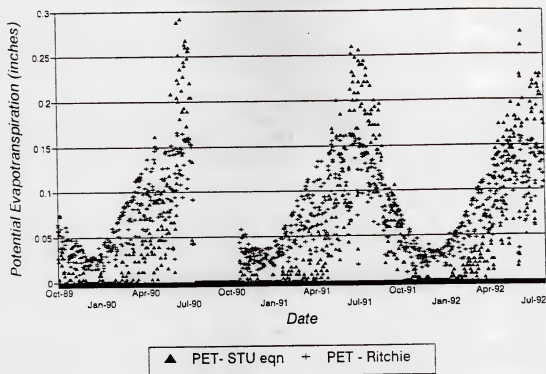


Figure 4.1.16. Comparison of estimated PET by the Ritchie and Solar Thermal Unit equations for Ramsay Flats.

4.1.3 Rainfall/Runoff Testing

Rainfall simulation tests measure the overall effect of each treatment due to both enhanced vegetation growth and to treatment effects on soil structure. The simulated rainfall/runoff tests were conducted between July 15-24, 1992 at sites 7, 21, and 33. At each test site, the rainfall simulation was applied to the instrumented replicate plot adjacent to the neutron access tube. The specific position of the testing apparatus around the tube was chosen to most closely represent the overall cover of the plot. All tests were originally proposed to be run at an application rate of 1.27 cm/hr (0.5 in/hr). However, an initial apparatus test conducted on the extra (unused) plot at site 21 strongly suggested that little or no runoff would be produced from most, if not all, plots at that application rate over the proposed test period of 2 hours. The initial test on the ag-till plot (15) was run at 2.54 cm/hr (1.0 in/hr) with most subsequent tests run at 3.8 cm/hr (1.5 in/hr). Application rates at site 7 were increased further to 5.1 cm/hr (2 in/hr). Ramsay Flats (site 21) was chosen for initial tests because it was thought to best facilitate testing methods.

The simulated storm event would be typical of an intense convective type storm with a recurrence interval in excess of the 100 year, 6 hour event (for all tested sites) and in excess of the 100 year, 24 hour event for all tests except the ag-till plot at site 21 (Miller et

al, 1973). The results of all runoff sample chemical analyses are located in Appendix Volume III, B4 and runoff/infiltration data are presented in Appendix Volume II, A2. The results of the rainfall/runoff tests are discussed by site in the following paragraphs.

The apparent erosion rate for each tested plot which produced runoff has been addressed by calculating the total mass of sediment from the weighted average of total suspended solids (TSS) in the two runoff samples collected during each test. Due to the small test area (0.36 m^2 , 3.8 ft^2) and that the tests were not replicated, the calculated rates should be used for plot comparisons only.

Neutron probe data was collected several times during each rainfall simulation test. These readings were collected prior to, during, and following the tests and were made in an effort to track the wetting front. The data were inconclusive because a portion of the monitored soil volume (less than half) was subject to wetting because the rainfall apparatus was set up adjacent to the access tube rather than directly over the tube. The choice of apparatus placement was driven by two factors: 1) the vegetation directly adjacent the monitoring sites (including the neutron probe access tubes) was generally not representative of the plot due to foot traffic, and 2) with the apparatus directly over the access tube, no readings could be obtained during the simulated rainfall tests without disrupting the applied rainfall.

Rocker: Tests were run on the deep plow, ag-till, injection and control plots at site 7. The coarse texture of the soils at this site severely limited runoff, with a total of 8.5 liters of runoff from the four tests and with no runoff from either the ag-till or deep plow plots (Figure 4.1.17). The cover soil treatment was not tested because the results of ag-till and deep plow tests at this site very strongly suggested that no runoff would be produced from the cover soil plot without modifying the simulation apparatus to greatly increase application rates. Application rates for the control plot were 3.8 cm/hr (1.5 in/hr) for 2 hours (no runoff produced) followed by an additional 2 hours at 5.3 cm/hr (2.1 in/hr). The latter application rate produced a total of 1 liter of runoff (0.3 cm or 0.11 in) by the end of hour 4. The deep plow and ag-till treatments received the 5.3 cm/hr rate for 2 hours without producing runoff (Table 4.1.8). The injection plot, also subjected to the 5.3 cm/hr application rate for 2 hours, produced a total of 7.5 liters of runoff. The runoff from this plot was likely enhanced by the apparent decrease in hydraulic conductivity of the surficial layer due to the large volume of lime slurry which was rejected from injection bores and/or broke through to the surface during the injection process. The edges of the plot had been diked during the injection process to prevent rejected material from running off the plot and this material was subsequently rototilled into the upper 20 cm (8 in) of the soil profile during seedbed preparation.

Data from the rainfall/runoff were used to compute SCS curve numbers for each tested plot using Equations [2] and [3] from Soil Conservation Service National Engineering Handbook, Section 4 - Hydrology, 1985.

$$Q = (P - I_a)^2 / ((P - I_a) + S) \quad [2]$$

where:

- Q = Total Runoff Volume (inches)
- P = Total Applied Precipitation (inches)
- I_a = Initial Rainfall Abstraction (inches) (the amount of rainfall applied prior to initiation of runoff)
- S = Maximum Potential Infiltration (inches)

$$S = \frac{1000}{CN} - 10 \quad [3]$$

where:

CN = SCS Curve Number

All parameters except S were derived from the rainfall simulation tests. Equation [2] was used to calculate S which was subsequently used in Equation [3] to solve for the curve number.

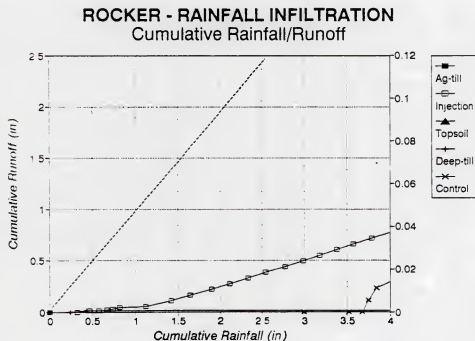


Figure 4.1.17. Cumulative rainfall and runoff from test plots at site 07 (Rocker).

Table 4.1.8. Rainfall/runoff test summary for site 07 (Rocker).

SCS CURVE NUMBERS					
SITE PLOT TREATMENT	TIME TO INITIAL RUNOFF (min)	RAINFALL RATE (in/hr)	TOTAL RAINFALL P (in)	TOTAL RUNOFF Q (in)	CN
07T13	NOT RUN	NOT RUN	NOT RUN	NOT RUN	NOT RUN
07D14	6.75	2.15	4.30	0.01	0.6
07A16	>120	2.08	4.15	0.00	NOT CALC
07C17	4.50	2.15	4.30	0.11	6.2 ¹
07I18	9.33	2.15	4.16	0.83	42.0

¹ Calculated using wet antecedent moisture conditions

The runoff data from site 07 strongly suggest that little runoff will be produced from this site during typical convective storm events, with or without treatment, but that the ag-till, deep plow and likely the cover soil treatments will essentially eliminate any site runoff. The high infiltration rates (very low curve numbers) and shallow ground water depths indicate most rainfall will infiltrate directly to ground water unless utilized by vegetation.

The pH and SC were generally measured for each 100 ml for the first 500 ml and then in 500 ml increments until runoff ceased following termination of the test. Runoff samples were obtained from only the control and injection treatments at this site (the 101 ml sample volume from the deep plow treatment was insufficient to analyze). The initial pH of runoff was similar for both the control and injection plots at 6.8 and 6.7 respectively (Figures 4.1.18). However, the pH of the injection plot runoff quickly climbed to about 8.5 and stabilized at that level for the duration of the test. The pH of the control plot runoff decreased to about 6.4 where it stabilized for the remainder of the test. The specific conductivity (SC) of injection plot runoff was approximately an order of magnitude greater than the control (Figure 4.1.18). The SC of control plot runoff stabilized at about 6 μ S after 300 ml of runoff was produced while the SC of the injection plot runoff decreased to about half of its initial value (72 μ S) after 3000 ml of runoff was produced. The data suggest that, while the injection plot SC was approximately an order of magnitude higher than the control plot runoff, runoff pH from the injection plot was about 2 units higher.

Runoff samples were collected and consisted of the initial 500 ml of runoff and a subsample from all remaining runoff (the composite sample). A comparison of total metal levels (Cu, Fe, Mn, and Zn) indicates that the injection treatment exhibited consistently higher levels of each metal in both the initial 500 ml sample and the overall composite sample. Dissolved metal analyses showed that concentrations of Cu, Fe, Mn, and Zn were decreased by the injection treatment while levels of calcium and magnesium were greatly increased. The injection treatment also decreased sodium concentrations and increased arsenic concentrations compared to the control. The decrease in sodium is likely due to the

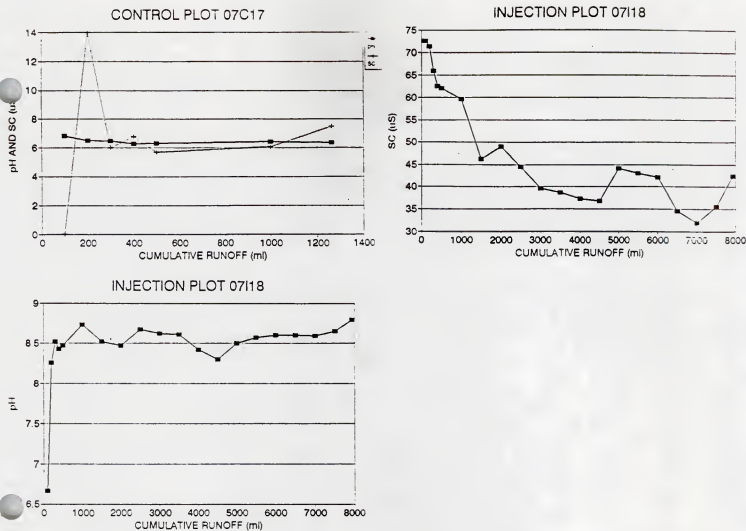


Figure 4.1.18. Specific conductance (SC) and pH of runoff from the Rocker injection and control plots.

displacement of sodium from exchange sites by calcium and subsequent flushing of sodium from the soil profile. The increase in arsenic, which is more mobile at neutral pH, may be due to a concurrent increase in pH. Initial and composite sample As concentrations were about 3 and 2 times higher respectively in the injection plot samples (52.8 vs 16.0 and 28.2 vs 11.5 $\mu\text{g/l}$ respectively). The treated plot runoff exhibited increased alkalinity in both initial (98 vs 3 mg/l) and composite samples (95 vs 22 mg/l); and reduced sulfate in runoff to below method detection limit (0.2 mg/l) from 1.5 and 0.8 mg/l found in respective samples from the control plot. The injection treatment had little effect on total suspended solids (TSS) in runoff. There was an apparent slight decrease in sodium in initial samples from the two plots (from 0.5 to 0.2 mg/l for control and injection plots respectively) and essentially no change in the composite levels for this anion (0.1 and 0.2 mg/l respectively). All runoff samples for the site exhibited fluoride concentrations below the 0.2 mg/l detection level. Nitrate concentrations were also very similar for the two plots, with both samples from the

control plot at 0.09 mg/l and both samples from the injection plot at 0.10 mg/l. Phosphate exhibited a slight increase in the injection plot versus the control with both control plot samples below the detection limit of 0.05 mg/l and the initial and composite samples from the injection plot at 0.08 and 0.07, respectively.

A comparison of total and dissolved analyses for copper, iron, manganese and zinc indicates most of the metal mass is either a component of or adsorbed onto the sediment. Concentrations of total metals were generally one to two orders of magnitude greater than dissolved constituents (Figures 4.1.19 and 4.1.20). It should be noted that the water quality of the limited runoff from site 07 is, in general, of much higher quality than runoff from either the Ramsay Flats (site 21) or Opportunity (site 33) sites.

The overall effect of the treatments was evaluated by combining the sample concentration data with the runoff volume data to determine the mass of constituents being transported from the test area (Figures 4.1.21 through 4.1.24). The data indicate increased mass of all parameters in runoff from the injection plot as compared to the control due to the greater runoff volume produced from the injection plot. This result is the effect of

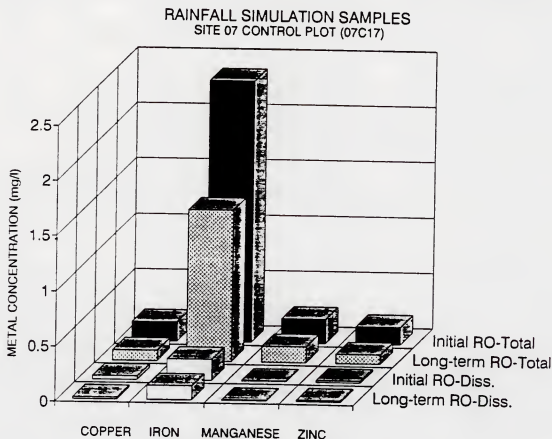


Figure 4.1.19. Dissolved and total metals in runoff from the Rocker control plot (07C17).

RAINFALL SIMULATION SAMPLES
SITE 07 INJECTION PLOT (07I18)

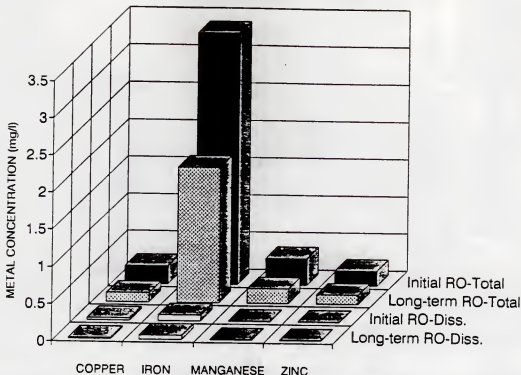


Figure 4.1.20. Dissolved and total metals in runoff from the Rocker injection plot (07I18).

decreased infiltration on the injection plot and the resulting increase in runoff as compared to the control.

The relative erosion potential from the site has been addressed by combining weighted TSS and runoff volumes to determine the total mass of sediment removed from the test area (Figure 4.1.25). No runoff, and therefore no sediment, was produced from the ag-till and deep plow plots. Sediment production from the control and injection plots was 90 and 510 mg respectively. This would be equivalent to 2.5 and 14 kg/ha or 2.2 and 12.7 pounds/acre respectively.

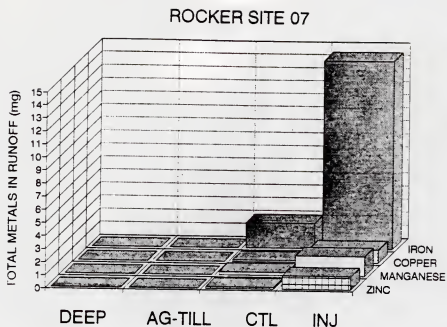


Figure 4.1.21. Mass of total metals in simulated rainfall runoff from site 07.

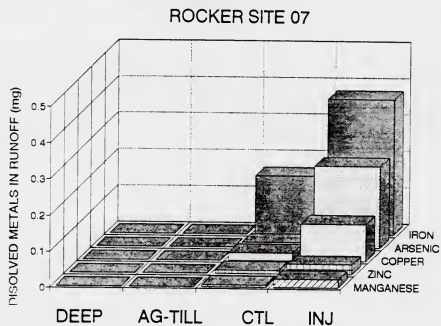


Figure 4.1.22. Mass of dissolved metals in simulated rainfall runoff from site 07.

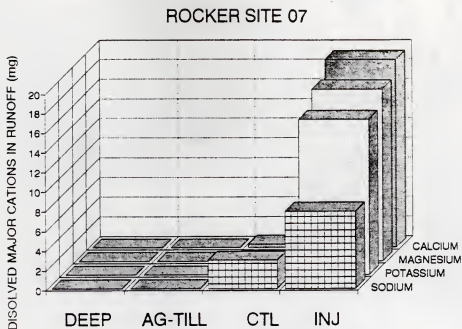


Figure 4.1.23. Mass of major cations in simulated rainfall runoff from site 07.

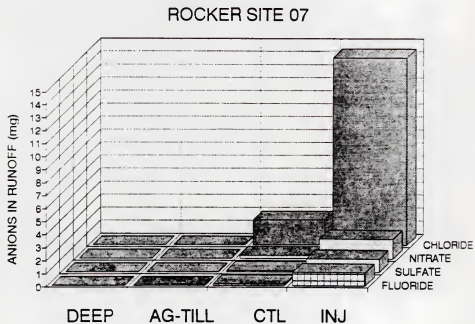


Figure 4.1.24. Mass of anions in simulated rainfall runoff from site 07.

ROCKER SITE 07

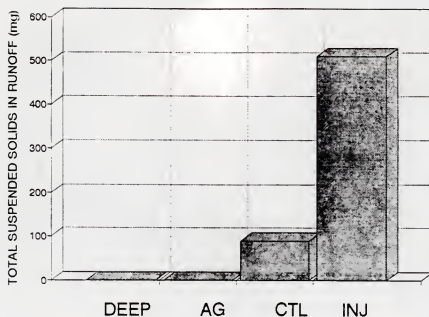


Figure 4.1.25. Mass of suspended sediment (TSS) produced in simulated rainfall runoff at site 07.

Ramsay Flats: The first actual test was conducted on the ag-till plot (15) using an application rate of 2.5 cm/hr (1.0 in/hr) with all remaining tests at this site run at 3.8 cm/hr (1.5 in/hr). Test results are summarized in Table 4.1.9.

Table 4.1.9. Rainfall/runoff test summary for Ramsay Flats site 21.

SITE PLOT TREATMENT	INITIAL RUNOFF TIME (min)	SCS CURVE NUMBERS			
		RAINFALL RATE (in/hr)	TOTAL RAINFALL P (in)	TOTAL RUNOFF Q (in)	CN
21T13	>120	1.00	3.00	0.02	2.2
21I14	10.58	1.00	3.01	1.78	87.0
21A15	14.73	1.00	1.98	0.31	55.6
21D17	10.08	1.50	2.98	0.77	59.0
21C18	10.75	1.50	2.99	2.23	94.3

Treatment effects on runoff were much more evident at site 21 compared to site 07 (Figure 4.1.26). Runoff ranged from near zero for the cover soil treatment (21T13) to over

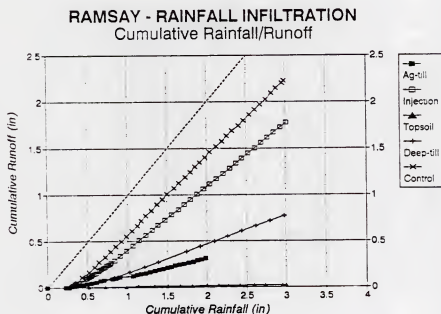


Figure 4.1.26.

Cumulative rainfall and runoff from test plots at Ramsay Flats site 21.

half of the applied rainfall on the control plot (21C18). Curve numbers varied from 2.2 to 94.3 for these treatments respectively. The ag-till was slightly more effective in reducing runoff than the deep plow treatment (possibly due to slightly greater slope of the deep plow test area produced by the plowing action) with the injection treatment only marginally better than the control plot. For comparison, curve numbers given for typical hydrologic soil-cover complexes are: hard surface roads, 74-92; fallow crop land, 77-94; and range (in good condition), 39-80 (Mockus, 1985).

Measurements of pH and SC in incremental runoff from site 21 indicate all treatments greatly reduced the SC of initial samples compared to the control plot (Figure 4.1.27). The deep plow plot (21D17) reduced the SC the most, followed by the ag-till and injection plots. Incremental pH values show that the injection plot produced the highest runoff pH with readings greater than 7 for most of the test. Deep plow plot runoff samples increased from pH 5.3 to 6.2 over the course of the test, runoff from the ag-till and control plots generally remained between pH 5 and 6 and below pH 5 respectively.

Total metals (Cu, Fe, Mn, and Zn) in runoff samples ranged 3 orders of magnitude, with initial samples from the control plot consistently exhibiting values an order of magnitude greater than any of the treated plots. The initial runoff sample, which contained 220, 170, 739, and 584 mg/l of Cu, Fe, Mn, and Zn respectively, demonstrates the potential of these untreated tailings materials to adversely affect receiving waters (Figure 4.1.28). Equally important is the high fraction of the total metal concentrations represented by the dissolved metals. Only iron exhibited a low dissolved fraction of the total portion in the samples. The ag-till plot runoff contained total concentrations of 19.5, 75.6, 23.7 and 26.6 mg/l for the respective metals in the initial sample (Figure 4.1.29). The concentrations fell to 10.4, 34.6, 11.0, and 14.2 mg/l for copper, iron, manganese and zinc in the composite sample. Dissolved metals represented a small fraction for all but manganese, for which the concentration was roughly half of the total concentration in both the initial and composite samples. Total metals in the injection plot runoff mimicked the ag-till but at somewhat reduced levels (Figure 4.1.30). Dissolved metal concentrations were also similar with the exception of manganese which was present at less than half the concentrations found in the ag-till samples. The deep plow plot runoff samples exhibited sharply reduced total concentrations of all metals except manganese with respect both the control and the other sampled treated plots (Figure 4.1.31). Manganese levels were comparable to the other treated plots. Deep plow plot runoff sample dissolved metal concentrations exhibited lower iron concentrations (32.3 and 17.7 $\mu\text{g/l}$ versus 77.2 and 55.7 $\mu\text{g/l}$ for ag-till, and versus 68.5 and 29.9 $\mu\text{g/l}$ for injection for initial and composite samples respectively). Arsenic concentrations were reduced from control levels in all samples from treated plots except the composite sample from the injection plot. The ag-till (36.1 and 40.4 $\mu\text{g/l}$ arsenic for initial and composite samples respectively) and deep plow treatments (48.7 and 47.3 $\mu\text{g/l}$ arsenic for initial and composite samples respectively) produced the least arsenic in runoff. Arsenic concentrations in injection plot runoff were comparable to control concentrations (116 and 91.6 versus 140 and 51.2 $\mu\text{g/l}$). The only notable trend in major cations (Ca, Mg, K, and Na) was a decrease in sodium in samples from all treated plots.

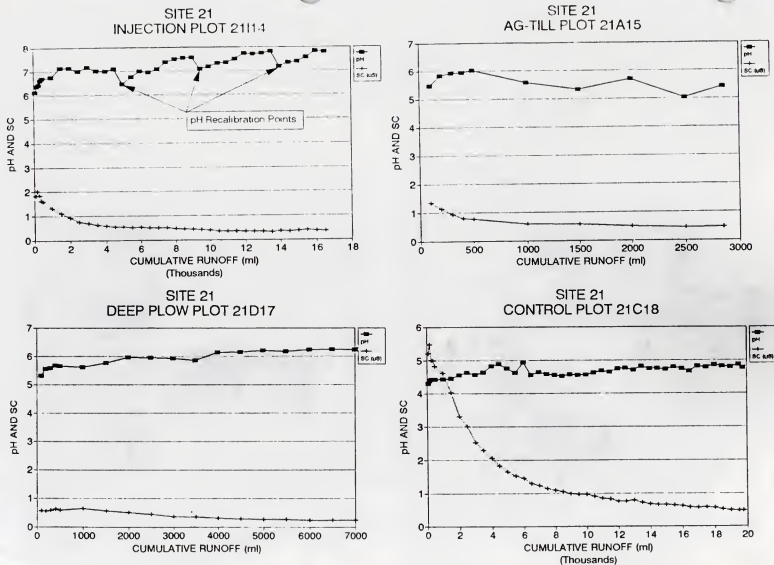


Figure 4.1.27. Specific conductance (SC) and pH of runoff from the Ramsay Flats injection, ag-till, deep plow, and control plots.

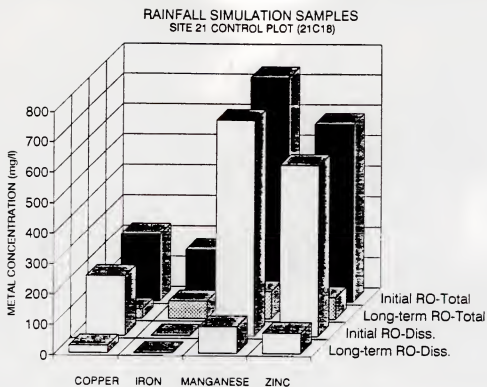


Figure 4.1.28.

Dissolved and total metals in runoff from the Ramsay Flats control plot (21C18).

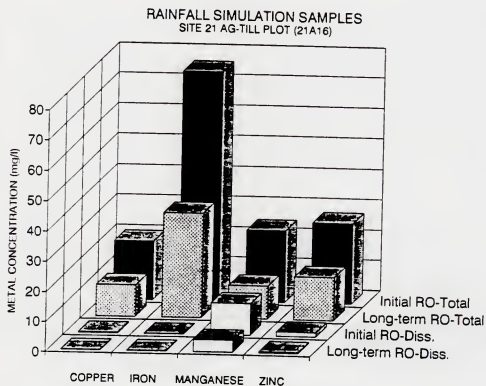


Figure 4.1.29.

Dissolved and total metals in runoff from the Ramsay Flats ag-till plot (21A15).

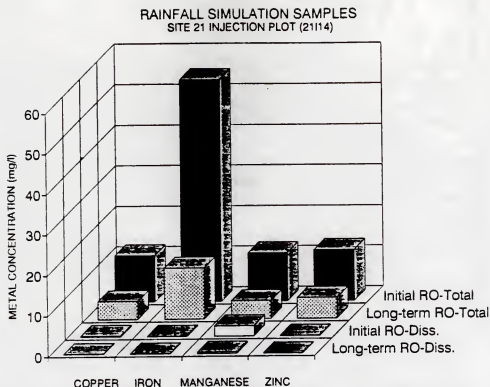


Figure 4.1.30. Dissolved and total metals in runoff from the Ramsay Flats injection plot (21I14).

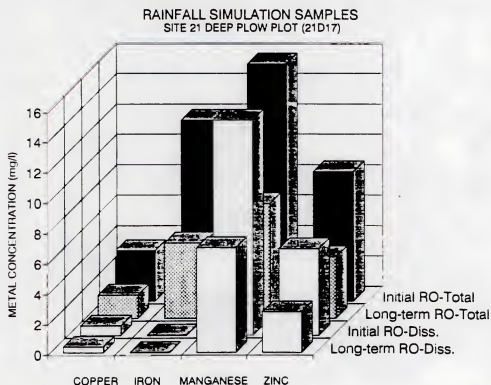


Figure 4.1.31. Dissolved and total metals in runoff from the Ramsay Flats deep plow plot (21D17).

Anions were dominated by sulfate in all samples which ranged from 3800 and 980 mg/l in initial and composite control samples to a low of 290 and 160 mg/l in deep plow samples. Chloride concentrations were also lower than control values (53 and 11 mg/l for initial and composite samples respectively) for all treated plots. The ag-till (3.1 and 0.9 mg/l) and deep plow (both samples 1.7 mg/l) produced the lowest chloride concentrations while the injection treatment chloride levels (12.0 and 1.8 mg/l) were about a quarter of the control concentrations. No definitive trends were apparent in fluoride levels which ranged from a maximum of 0.8 mg/l in the initial ag-till plot sample to 0.2 for the composite sample from the injection plot. Nitrate concentrations were greatest for control samples (38 and 6.3 mg/l), and lowest in the deep plow (1.2 and 0.3 mg/l). The ag-till (3.2 and 0.7 mg/l) and injection (10.0 and 1.3 mg/l) were still well below the control concentrations. Phosphate levels were above the detection concentration for only the injection plot (0.11 and 0.06 mg/l) and the composite sample from the control (0.06 mg/l).

The calculated mass of total and dissolved metals, major cations and anions in runoff exhibits the overall effectiveness of the treatments (Figures 4.1.32 through 4.1.35). The cover soil treatment is apparently the best treatment employed at Ramsay Flats for the purpose of limiting off site migration of surface metals. This treatment eliminated any runoff from the 3 inch, 2 hour applied storm. The ag-till and deep plow were about equally effective in reducing overall surface metal transport at the site and the injection was somewhat less effective but still exhibited a substantial reduction from control levels.

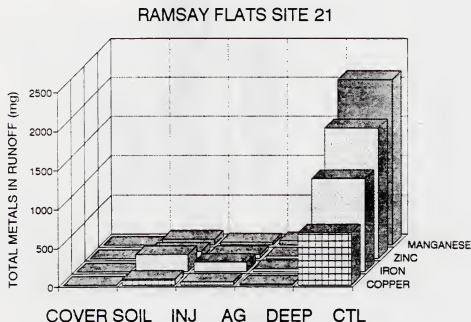


Figure 4.1.32. Mass of total metals in simulated rainfall runoff from site 21.

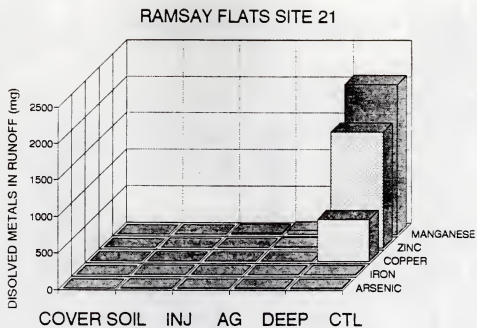


Figure 4.1.33. Mass of dissolved metals in simulated rainfall runoff from site 21.

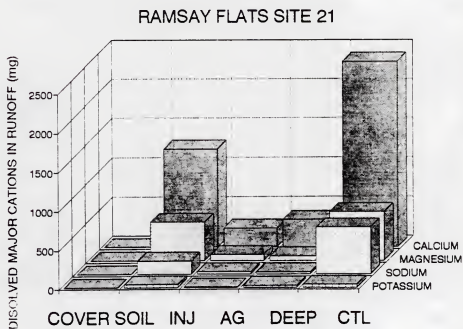


Figure 4.1.34. Mass of major cations in simulated rainfall runoff from site 21.

Erosion rates, as represented by the total suspended solids data, indicate an identical order of effectiveness. Sediment production was control > injection > deep plow > ag-till > cover soil (Figure 4.1.36).

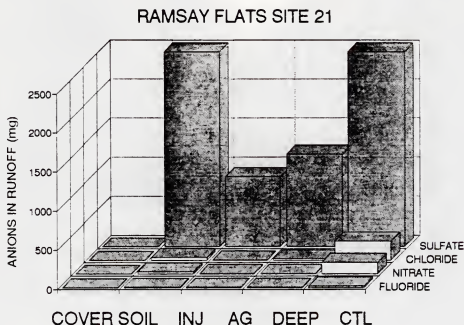


Figure 4.1.35

Mass of anions in simulated rainfall runoff from site 21.
RAMSAY FLATS SITE 21

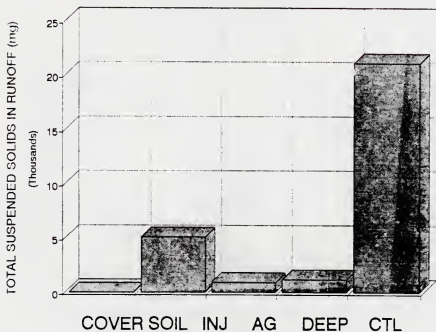


Figure 4.1.36.

Mass of suspended sediment (TSS) produced in simulated
rainfall runoff at site 21.

Opportunity: Rainfall simulation tests were run using a 3.8 cm/hr (1.5 in/hr) application rate on the five instrumented plots at the Opportunity site (Figure 4.1.37). The application rate was insufficient to produce runoff from either the cover soil or deep plow plots (Table 4.1.10). Runoff from the remaining plots ranged from 1.5 cm (0.58 in) for the ag-till treatment to 6.1 cm (2.42 in) for the control. Calculated curve numbers were 58.2, 66.3, and 93.2 for the ag-till, injection, and control plots respectively. The CN for the control would be typical of hard surfaced roads or fallow crop land while the remaining values are within the range for pasture or native range. These data suggest the cover soil and deep plow treatments were very effective in eliminating runoff while the ag-till treatment greatly reduced runoff. The injection treatment, while less effective than the cover soil, deep plow or ag-till, still reduced runoff to 43 percent of that observed for the control.

The specific conductance and pH of incremental samples from the 3 plots which produced runoff at this site (ag-till, injection and control) indicated both treated plots exhibited lower initial SC values and much higher pH values compared to the control (Figure 4.1.38). The SC of all runoff generally decreased quickly to stable levels. These data suggest that the treated plots have reduced the accumulation of salts on the soil surface and likely have reduced the dissolved portion of total metals in the runoff.

The six samples collected from the ag-till, injection, and control plots indicate marked differences in both total and dissolved metals between runoff from the control plot and both treated plots producing runoff (Figures 4.1.39 through 4.1.41).

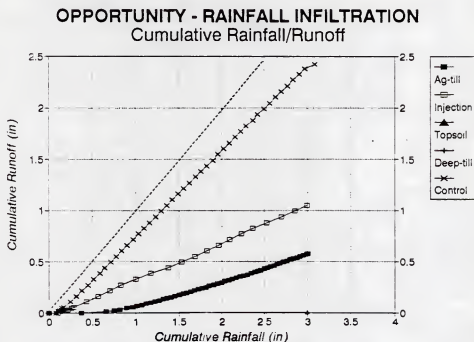


Figure 4.1.37.

Cumulative rainfall and runoff for the Opportunity site (site 33).

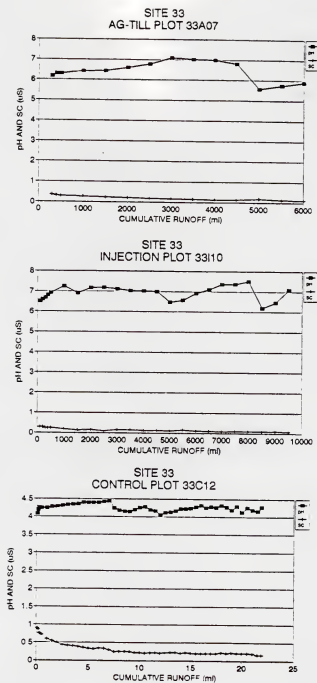


Figure 4.1.38. Specific conductance (SC) and pH of runoff from the Ramsay Flats injection, ag-till, deep plow, and control plots.

Table 4.1.10.

Rainfall/runoff test summary for site 33 (Opportunity).

SCS CURVE NUMBERS					
SITE PLOT TREATMENT	INITIAL RUNOFF TIME (min)	RAINFALL RATE (in/hr)	TOTAL RAINFALL P (in)	TOTAL RUNOFF Q (in)	CN
33A07	26.30	1.50	3.00	0.00	58.2
33D08	>120	1.50	3.00	0.00	NOT CALC
33T09	>120	1.50	3.00	0.00	NOT CALC
33I10	4.33	1.50	3.00	1.05	66.3
33C12	3.28	1.50	3.09	2.42	93.2

Total copper was reduced from 60 and 14.3 mg/l observed in control plot samples to 1.3 and 0.6 mg/l in the ag-till plot samples and 3.4 and 1.0 mg/l in the injection plot samples for initial and composite samples respectively. These trends were consistent for Fe, Mn, and Zn. The dissolved portion of total metals for the control plot was 50 and 57 percent for Cu, 1.4 and 0.4 percent for Fe, 79 and 84 for Mn, and 76 and 82 percent for Zn in initial and composite samples respectively. Similar values for the ag-till plot were 0.03 and 16 percent for Cu, 0.9 and 1.5 percent for iron, 73 and 31 percent for manganese, and 33 and 8.1 percent for Zn. The percentage of total metals represented by the dissolved fraction was

RAINFALL SIMULATION SAMPLES
SITE 33 (OPPORTUNITY) AG-TILL PLOT

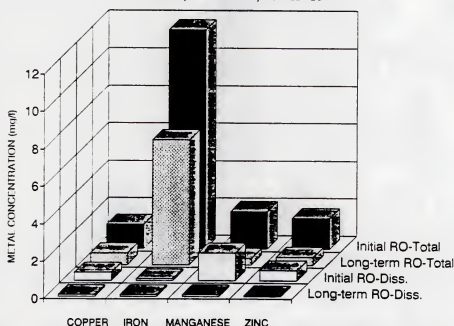


Figure 4.1.39.

Dissolved and total metals in runoff from the Opportunity ag-till plot (33A07).

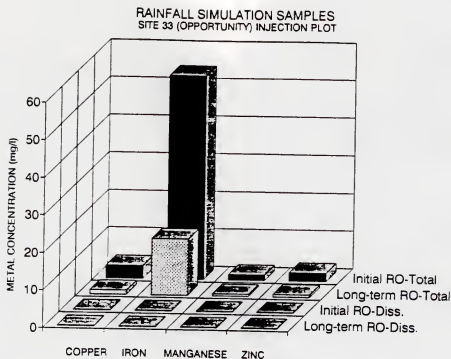


Figure 4.1.40.

Dissolved and total metals in runoff from the Opportunity injection plot (33I10).

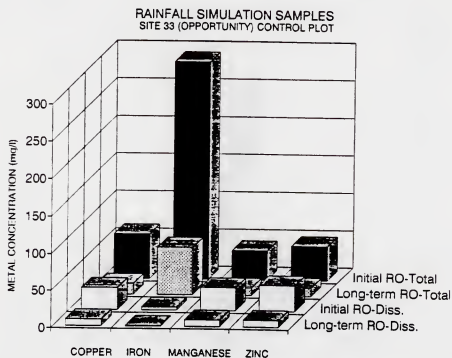


Figure 4.1.41.

Dissolved and total metals in runoff from the Opportunity control plot (33C12).

lowest for the injection plot samples: 2.5 and 3.1 percent for Cu, 0.3 and 0.6 percent for iron, 3.4 and 2.8 percent for manganese, and 0.9 and 1.8 percent for Zn in initial and composite samples respectively. Runoff from treated plots exhibited an increase in Ca and decreases in both Mg and Na compared to the control. Potassium concentrations did not show any conclusive trends. The highest K concentration was found in the initial ag-till sample (9.61 mg/l) and the lowest concentration was found in the injection plot composite sample (0.61 mg/l).

Arsenic values ranged from a high of 34.4 $\mu\text{g/l}$ to a low of 3.6 $\mu\text{g/l}$ for the initial and composite control plot samples. There was little difference between treated plots with 15.6 and 9.8 $\mu\text{g/l}$ for the ag-till plot samples and 15.3 and 10.7 $\mu\text{g/l}$ for the injection plot samples for initial and composite respectively. Alkalinity ranged from below the detection limit (2 mg/l) for both control samples to 102 and 31 mg/l for the initial and composite ag-till samples. Injection plot alkalinity concentrations were intermediate with 72 mg/l for the initial sample and 31 mg/l for the composite sample.

Sulfate dominated anion concentrations in all samples. Initial and composite samples for the control, ag-till, and injection plots were: 490, 130; 130, 61; 160, and 48 mg/l respectively. These values represent a 3 to 4 fold decrease in sulfate in initial samples and a 2 fold decrease in composite concentrations for treated plots compared to the control. Chloride concentrations varied from 7.9 mg/l (initial control sample) to 0.5 mg/l (composite injection sample). Chloride levels in treated plot samples were 2 to 10 times below control sample concentrations with the injection plot exhibiting the lowest values (0.7 and 0.5 mg/l). Fluoride was found above detection concentrations for only both control plot samples (1.6 and 0.6 mg/l) and the initial injection plot sample (0.3 mg/l). Nitrate concentrations were greatest for the control plot (3.5 and 0.7 mg/l), with all treated plot samples equal to or less than 0.4 mg/l. Only one sample (initial injection plot sample at 0.06 mg/l) exhibited phosphate concentrations above the detection limit (0.05 mg/l).

The mass of total metals, dissolved metals, major cations, and anions transported from the rainfall simulation test sites are presented in Figures 4.1.42 through 4.1.45. The control plot produced the greatest mass of all parameters while the cover soil and deep plow plots produced the least (none). In general, the ag-till treatment preformed slightly better than the injection treatment but there was less difference between these treatments at site 33 than observed at site 21. This trend was also apparent in sediment production (Figure 4.1.46). Sediment production for the ag-till, injection, and control treatment plots was 10.0, 14.4 and 85.0 kg/ha (8.9, 12.8, 75.7 lbs/ac) respectively.

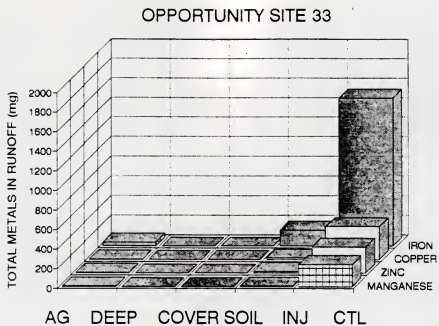


Figure 4.1.42. Mass of total metals in simulated rainfall runoff from site 33.

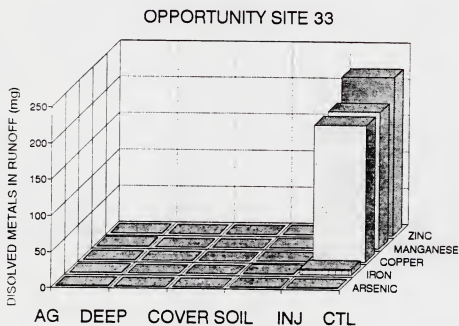


Figure 4.1.43. Mass of dissolved metals in simulated rainfall runoff from site 33.

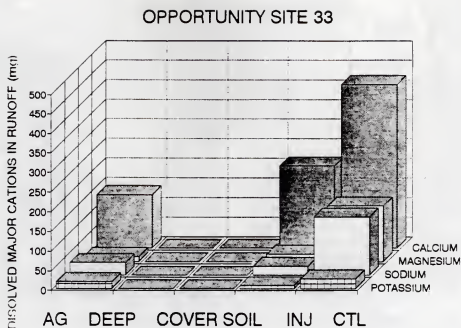


Figure 4.1.44. Mass of major cations in simulated rainfall runoff from site 33.

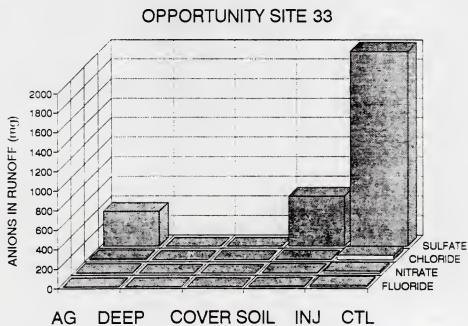


Figure 4.1.45. Mass of anions in simulated rainfall runoff from site 33.

RAINFALL/RUNOFF TESTS

OPPORTUNITY SITE 33

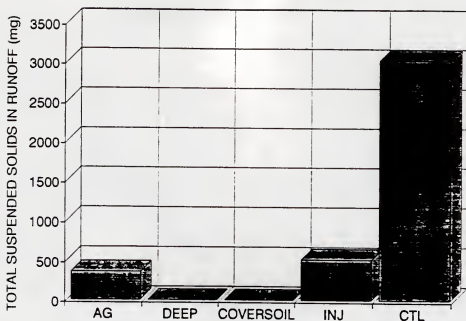


Figure 4.1.46. Mass of suspended sediment (TSS) produced in simulated rainfall runoff at site 33.

Rainfall Simulation Test Summary: The rainfall simulation tests indicate all implemented treatments have some ability to reduce runoff volumes and to improve the water quality of that runoff from the finer textured soils at sites 21 and 33. The effects of treatments were less apparent at site 07 where coarse textured soils promoted infiltration. Since these tests were not replicated, no statistical significance can be determined from the existing data. However, some general trends are clearly evident. The cover soil treatment would appear to be the most effective of those tested in that it eliminated runoff under the rainfall application rate tested (1.5 in/hr) for 2 hours at both sites 21 and 33 (the cover soil treatment was not tested at site 07). The deep plow treatment was nearly as effective in that only one of three tested plots produced runoff (Ramsay Flats). Overall, the ag-till was generally less effective than the deep plow but still reduced runoff markedly and improved runoff quality compared to the control. At site 21, this treatment produced less runoff than the deep plow treatment, but the effect may have been the result of increased slope on this deep plow plot resulting from the plow "furrows". The injection treatment actually resulted in increased runoff (as compared to the control plot) at site 7. However, the increase in the quality of runoff still produced a reduction of off-site transport of all parameters analyzed. The injection treatment was relatively effective in reducing dissolved and total metal levels in runoff at all sites. It was ineffective in reducing sulfate concentrations (compared to the control) at site 21. This treatment was nearly as overall effective as the ag-till treatment at site 33.

Specific site differences were readily evident. At site 7, most applied rainfall on both treated plots and the control plot apparently infiltrated into the coarse textured soils. The calculated curve number for the control plot at this site was very low (6.2). This suggests that most precipitation falling at site 07, if not utilized by evapotranspiration demands, will percolate to the groundwater system. At site 21, the untreated tailings curve number (94) was similar to that expected for hard surfaced roads and most precipitation could be expected to contribute to runoff. All applied treatments were at least somewhat effective in reducing the runoff potential, with the ag-till and deep plow plots exhibiting curve numbers similar to range in good condition. The cover soil treatment essentially eliminated runoff at site 21. These findings suggest that a large portion of the precipitation falling on untreated tailings at site 21 will runoff and that all STARS treatments, except the slurry injection, will markedly decrease potential runoff at the site. At site 33, conditions were apparently similar to site 21 with respect to the relative volume of runoff produced. The control plot again exhibited a curve number typical of hard surfaced roads while both the cover soil and deep plow treatments eliminated runoff at the site under the applied rainfall rates. Runoff from the ag-till and injection plots was similar to typical range conditions.

From a rainfall/runoff perspective, the cover soil, deep plow and the ag-till treatments show a high potential for reducing runoff from intense convective type storms with 100 year return intervals and the ability to greatly improve the quality of runoff that is produced. Other considerations, such as costs and implementability (Sections 4.4 and 4.5), will likely drive the final treatment selection for application on a broad scale.

4.2 SOIL CHEMISTRY MONITORING

Soils and mine wastes along Silver Bow Creek have become a potential source of inorganic contaminants resulting from historic mining, milling and smelting of metalliferous ores in Butte. The potential for metals contained in streambank soils and sediments to contaminate other media depends on the relative mobility of each of the chemical "partitions" in which the metals are contained. Metal contaminants are held in one or more of the following waste/soil partitions in order of decreasing mobility: 1) dissolved in soil solution; 2) bound to exchange sites of organic or inorganic particles; 3) precipitated with other compounds in waste/soil; 4) incorporated into biological material; and 5) occluded into waste/soil minerals. Metals in the first two fractions are mobile and bioavailable - metal in the last three are considered immobile, but can become mobile under certain circumstances. Kinetics controlling chemical equilibrium of each metal contaminant between the solution phase and the solid phase are not well understood. Significant factors include pH, oxidation reduction potential (Eh), amount of organic matter, clay mineralogy, abundance of oxides of iron, aluminum and manganese, and presence of soluble chelators. The relative abundance of metals in various phases is routinely investigated using various analytical techniques (eg. total analysis by x-ray fluorescence) and by employing soil extracts such as distilled water and ammonium acetate.

The various STARS treatments may have significantly altered the equilibrium between the mobile and immobile fractions for each metal contaminant. Soil chemical analyses were used to measure the short-term effectiveness of various STARS treatments in neutralizing acidity, and controlling bioavailability and mobility of the metal contaminants. Six soil chemistry monitoring activities were performed:

- Monitoring pH and electrical conductivity (EC) to assess tailings neutralization and soluble salt concentrations;
- Measuring water soluble metal and arsenic levels to assess potential contaminant movement to groundwater;
- Sampling soil pore water *in-situ* to measure the abundance of metals;
- Measuring extractable (bioavailable) metal and arsenic levels to assess potential for bioaccumulation in STARS vegetation;
- Measuring surface soil metal and arsenic concentrations within and outside the STARS plots to assess treatment effects in changing contaminant movement via dust and direct contact pathways; and
- Measuring degree of amendment mixing into the mine wastes to assess effectiveness of the various STARS incorporation techniques.

4.2.1 Water Soluble Metals, pH and EC

In the spring of 1990, a truck-mounted Giddings core barrel sampler was used to collect amended wastes/soils from every experimental plot. Depth increments collected included 0-8 cm and 8-15 cm layers (these corresponded to the shallow amendment incorporation depths employed in the agricultural tillage treatment and beneath the constructed coversoil wedge treatment); the 15-30, 30-60, 60-90 and 90-120 cm layers (these corresponded to amended tailings and amended natural soils within the deep incorporation zone); and the 120-150 and 150-180 cm layers (these were below the amendment incorporation zone). The deep samples were collected in the deep plow, slurry injection, and control plots. Soil samples were also obtained from within the coversoil wedge treatments at two depth increments (0-23 and 23-46 cm).

Collected samples were air-dried and sieved to obtain the ≤ 2 mm fraction. A saturated paste of each sample was prepared and the pH and EC levels in the extract solution were measured. Figures 4.2.1 through 4.2.3 exhibit the mean pH of the material from each depth increment arranged by STARS treatment. At Site 2, Manganese Stockpile (Figure 4.2.1, top), the pH values of the materials in the treated plots were much higher than the control plot wastes, especially for the upper depth increments. At a tailings depth of approximately 40 cm, wastes treated by agricultural tillage revealed a similar mean pH to the untreated materials. In contrast, wastes amended with lime using the two deep incorporation techniques exhibited elevated mean pH levels throughout the profile.

At Site 7, Rocker (Figure 4.2.1, bottom), mean pH levels of the upper amended wastes layers (0-15 and 15-30 cm) were elevated compared to the mean value of the control plots. Low pH levels were encountered below the zone of incorporation for the agricultural tillage treatment. At the deepest profile depths, the pH of the control, deep plow and injected treatment plots were all similar. These depths were below the amendment incorporation zone and the relatively high pH values most probably indicate these materials were natural soils - not effected by the overlaying mine waste materials. Similar patterns of pH, depth of treatment, and depth to natural soils were exhibited at the other three STARS sites (Figures 4.2.2 and 4.2.3).

The impact of the STARS treatments on the measured electrical conductivity of the amended wastes/soils are shown in Figures 4.2.4 through 4.2.6. Electrical conductivity is a measure of the soluble salts in the materials. Both the concentration of the soluble salts and their identity may have significant impacts on the revegetation potential of the amended materials. The mean EC levels of amended materials at Site 2 are shown in Figure 4.2.4, top. In the upper profile depths, the EC values were elevated compared to the control plot values. Below the zones of amendment incorporation, the measured EC levels of materials in the treated plots were similar to the wastes/soils in the control plots. These two trends were repeated for amended wastes/soils at the other STARS sites as exhibited in Figures 4.2.5 through 4.2.6.

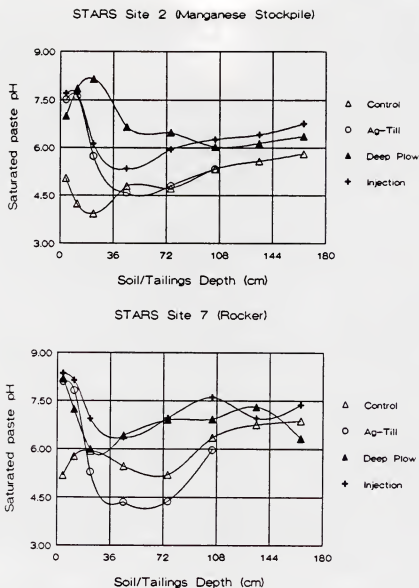


Figure 4.2.1. Mean pH for amended wastes/soils at STARS Site 2-Manganese Stockpile (top) and STARS Site 7-Rocker (bottom).

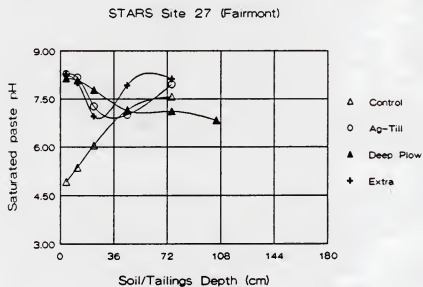
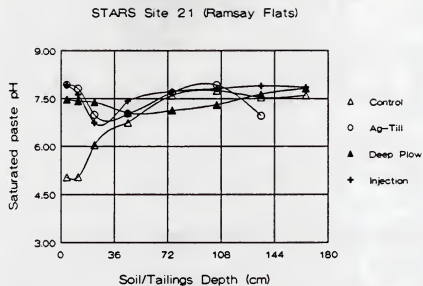


Figure 4.2.2. Mean pH of amended wastes/soils at STARS Site 21-Ramsay Flats (top) and STARS Site 27-Fairmont (bottom).

STARS Site 33 (Opportunity)

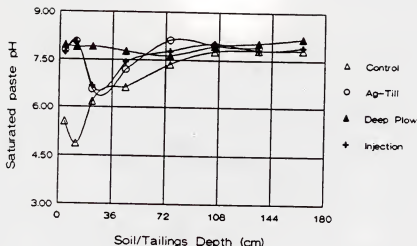
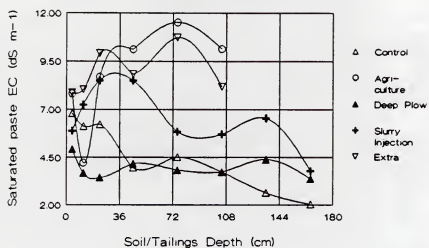


Figure 4.2.3. Mean pH of amended wastes/soils at STARS Site 33-Opportunity.

Saline soil refers to a soil that contains sufficient soluble salts to impair its productivity. The salt content of soils above which vegetation growth is affected depends on the composition of the salt, distribution of the soluble salts in the soil profile, soil texture, and plant species. The presence of white crusts of salts on the soil surface is often indicative of saline soils. The use of calcium carbonate, calcium hydroxide and the other amendments in the STARS treatments not only affected pH, but also changed the composition of the soluble salts. Metal contaminants (copper, zinc, cadmium, lead, etc.) were removed from the soil solution and were replaced by calcium. As a result, the pH of the amended wastes was ameliorated and the metal contaminants in the soil solution were replaced by a less phytotoxic cation - calcium. These concepts are presented in detail in the pore water chemistry (Section 4.2.2).

As part of the extractable soil chemistry monitoring (see Section 4.2.3), samples of amended wastes/soils were collected in 1991 from two depths, 0-15 and 15-46 cm. In addition to extractable chemistries, these samples were also analyzed for pH and EC levels in prepared saturated paste extract solutions. The purpose for obtaining this information was to detect any changes in pH or EC in the amended wastes/soils from 1990 to 1991. Electrical conductivity and pH data from identical plots within each site were paired and depth integrated values were calculated for the 0-15 cm depth layer. Data from 1990 were then compared to 1991 using the paired t-test. Results are presented in Table 4.2.1. No significant changes were found in pH levels of the amended wastes/soils at the four STARS sites. Electrical conductivity levels were significantly elevated in 1991 for materials at Site 7 (Rocker) compared to the 1990 data.

STARS Site 2 (Manganese Stockpile)



STARS Site 7 (Rocker)

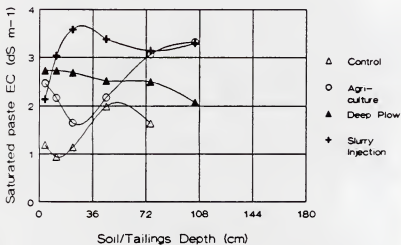
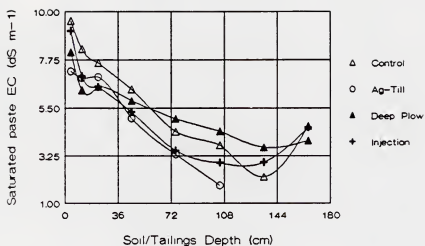


Figure 4.2.4. Mean EC of amended wastes/soils at STARS Site 2-Manganese Stockpile (top) and STARS Site 7-Rocker (bottom).

STARS Site 21 (Ramsay Flats)



STARS Site 27 (Fairmont)

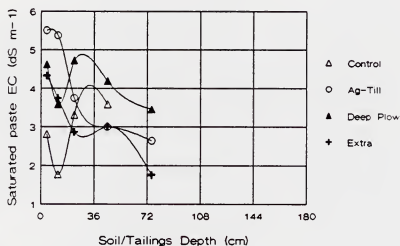


Figure 4.2.5. Mean EC of amended wastes/soils at STARS Site 21-Ramsay Flats (top) and STARS Site 27-Fairmont (bottom).

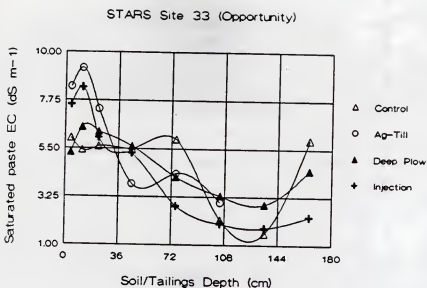


Figure 4.2.6. Mean EC of amended wastes/soils at STARS Site 33-Opportunity.

Concentrations of soluble elements were measured in the amended wastes/soils. The collected samples were composited to reduce the total number of samples. For example, the 0-8 cm and 8-15 cm layers from each Giddings core were composited. In a like manner, the 15-30 and 30-60 cm layers were composited. This activity was conducted by core so that the integrity of each experimental plot was maintained. Saturated paste extract solutions were prepared and the concentration of several metals and As were determined in the extractant solutions. Mean levels of water soluble metals in the amended wastes/soils are shown in Tables 4.2.2 through 4.2.6. Data are arranged by profile depth, STARS treatment, and element. Data for the coversoil wedge treatment are for materials beneath the wedge. The mean values were compared using Analysis of Variance (ANOVA) and were separated based on least significant difference (LSD) at $p \leq 0.05$. The expected zone(s) of amendment incorporation is also indicated in the table. The additions of the amendments significantly reduced the levels of soluble Cd, Cu, and Zn in the treated plots compared to the control for the 0-15 cm depth at Site 2 (Table 4.2.2).

Some treatments (deep plow) reduced the concentration of As in the 0-15 cm depth, while soluble Pb levels were not effected by the STARS treatment at this depth. Soluble metal concentrations at the 0-15 cm layer are graphically displayed in Figure 4.2.7, top. These data are arranged by treatment and the concentration means have been transformed using \log_{10} to accommodate the relative levels. Information in this figure indicate dramatic reductions in soluble Cd, Cu, and Zn concentrations by the STARS treatments, and only

Table 4.2.1. Comparison of changes in STARS soil/amended waste pH and EC between 1990 and 1991 from saturated paste extract.

	Site 7	Site 21	Site 27	Site 33
Mean pH 1990	7.55	7.19	7.16	7.12
Mean pH 1991	7.54	7.03	7.51	7.08
N ¹	10	10	8	10
p ²	0.50	0.25	0.16	0.62
Mean EC 1990	1.31	9.17	3.59	7.20
Mean EC 1991	1.91	7.85	4.33	5.83
N ¹	10	10	6	10
p ²	0.05	0.30	0.12	0.18

¹ N is the number of paired comparisons across all treatments.

² p values derived using parametric paired t-test, 2-tailed distribution.

slight changes in As and Pb concentrations. At deeper amendment incorporation depths, some contaminant levels were reduced while others were similar to the levels found in the control plots. In general, the waste materials in which the amendments were incorporated using the deep plow revealed the greatest reduction of soluble metals through the profile.

At STARS Site 7 (Rocker), there was insufficient data to complete the ANOVA. Levels of soluble metals were apparently reduced by the addition of the amendments compared to the control plot concentrations (Table 4.2.3).

Mean concentrations of water soluble metals and As in amended wastes/soils at STARS Site 21 are displayed in Table 4.2.4. Concentrations of Cd, Cu, and Zn were significantly reduced in the 0-15 cm zone compared to the control mean levels (Figure 4.2.7, bottom and Table 4.2.4). Arsenic concentrations were, however, significantly enhanced indicating an increased mobility of As in this profile depth. This increased As mobility is clearly exhibited in Figure 4.2.8, top. Only slight (nonsignificant) increases in As levels were found in the lower profile depths. In contrast to the increase As availability, the decreased concentration of soluble Cu is displayed in Figure 4.2.8 (bottom).

Mean levels of water soluble metals and As in amended wastes/soils at STARS Site 27 are exhibited in Table 4.2.5. Changes in concentrations via treatment were not as obvious at this site. There appeared to be a trend of enhanced As mobility in the 0-15 cm layer but

Table 4.2.2. Mean concentrations¹ of water soluble metals ($\mu\text{g/kg}$) in amended wastes/soils at STARS Site 2.

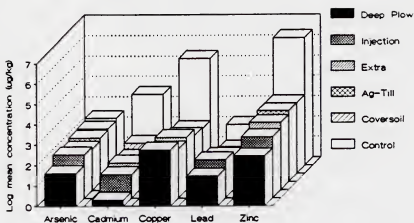
Amended Waste/ Soil Depth (cm)	Treatment					
	Control	Ag Till	Deep Plow	Injection	Coversoil	Extra
--- Arsenic ---						
0-15	211 B ²	140 AB	41 A	132 AB	157 AB	102 AB
15-60	180 A	162 A	105 A	65 A	76 A	55 A
60-120	91 A	132 A	80 A	41 A	140 A	214 A
120-180	77 A	--	29 A	15 A	--	--
--- Cadmium ---						
0-15	3133 C	16 B	2 A	14 AB	31 B	21 B
15-60	5432 CB	1489 B	9 A	17298 C	743 B	6109 CB
60-120	8710 A	13490 A	698 A	3412 A	638 A	1660 A
120-180	1585 A	--	106 A	863 A	--	--
--- Copper ---						
0-15	184077 B	183 A	547 A	246 A	156 A	361 A
15-60	245471 C	75162 CB	344 A	75509 CB	25763 B	55081 CB
60-120	31989 BC	216770 C	5152 AB	953 A	162555 C	8933 AB
120-180	6281 A	--	2518 A	700 A	--	--
--- Lead ---						
0-15	99 A	21 A	28 A	71 A	17 A	47 A
15-60	186 A	242 B	32 A	301 B	219 B	120 AB
60-120	454 ABC	1250 C	159 AB	515 ABC	92 A	634 BC
120-180	1019 A	--	95 A	163 A	--	--
--- Zinc ---						
0-15	1794734 B	2897 A	267 A	933 A	2838 A	1849 A
15-60	1127197 B	623735 B	635 A	1380384 B	246604 B	1174898 B
60-120	505825 AB	1770109 B	85310 A	232809 AB	234963 AB	1663413 B
120-180	311889 A	--	30539 A	53211 A	--	--

¹ Geometric mean values are based on N = 3 or 4.

² Multiple mean comparison based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

Indicates depth of amendment incorporation.

STARS Site 2 (Mn Stockpile, 0-15 cm)



STARS Site 21 (Ramsay Flats, 0-15 cm)

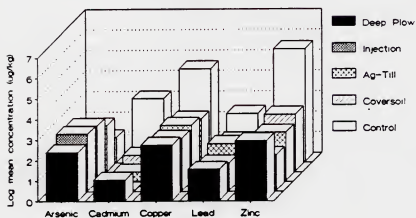


Figure 4.2.7. Mean water soluble metal levels (0-15 cm) for STARS Site 2 (Manganese Stockpile) and STARS Site 21 (Ramsay Flats).

Table 4.2.3. Mean concentrations¹ of water soluble metals ($\mu\text{g/kg}$) in amended wastes/soils at STARS Site 7.

Amended Waste/ Soil Depth (cm)	Treatment				
	Control	Ag Till	Deep Plow	Injection	Coversoil
--- Arsenic ---					
0-15	46	8.8	—	—	37
15-60	25	4.3	20	45	11
60-120	7.1	7.9	7.2	189	8.3
120-180	9.3	—	1.4	4.5	—
--- Cadmium ---					
0-15	8.7	0.8	—	—	1.7
15-60	14	—	6.3	7.4	30
60-120	210	45	14	6.2	296
120-180	99	—	74	1.4	—
--- Copper ---					
0-15	613	134	—	—	64
15-60	1800	3819	428	274	785
60-120	18281	285	270	189	2070
120-180	752	—	87	14	—
--- Lead ---					
0-15	99	12	—	—	6.1
15-60	17	32	37	71	26
60-120	429	63	14	32	54
120-180	337	—	2.6	5.6	—
--- Zinc ---					
0-15	1781	53	—	—	36
15-60	1874	18376	832	2070	3890
60-120	58614	29580	1636	199	51404
120-180	10500	—	20500	3440	—

¹ Geometric mean values are based on N = 1, 2, 3 or 4 observations.

Indicates depth of amendment incorporation.

Table 4.2.4. Mean concentrations¹ of water soluble metals ($\mu\text{g/kg}$) in amended wastes/soils at STARS Site 21.

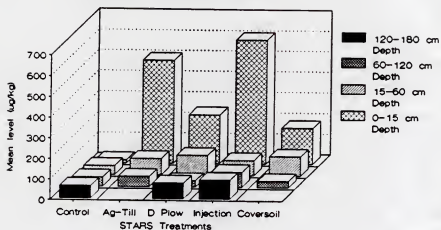
Amended Waste/ Soil Depth (cm)	Treatment				
	Control	Ag Till	Deep Plow	Injection	Coversoil
--- Arsenic ---					
0-15	16 A ²	497 B	236 B	600 B	175 B
15-60	44 A	79 A	97 A	73 A	94 A
60-120	45 A	50 A	31 A	73 A	31 A
120-180	61 A	—	16 B	92 A	—
--- Cadmium ---					
0-15	1183 C	2.9 AB	10 B	1.1 A	6.3 B
15-60	489 A	53 A	88 A	79 A	60 A
60-120	3.6 A	2.4 A	49 B	4.8 A	4.9 AB
120-180	3.1 A	—	2.6 A	2.8 A	—
--- Copper ---					
0-15	33419 B	526 A	516 A	370 A	353 A
15-60	3972 B	910 AB	938 AB	324 A	1312 AB
60-120	60 AB	41 A	589 C	127 B	138 B
120-180	58 A	—	93 A	95 A	—
--- Lead ---					
0-15	218 A	68 A	88 A	23 A	34 A
15-60	399 A	124 A	198 A	122 A	56 A
60-120	4.9 A	5.5 A	127 B	23 AB	21 AB
120-180	10 A	—	8.9 A	6.5 A	—
--- Zinc ---					
0-15	303389 C	235 AB	815 B	56 A	579 B
15-60	91201 A	6471 A	11429 A	8790 A	6637 A
60-120	249 A	74 A	7834 B	159 A	682 AB
120-180	327 A	—	173 A	67 A	—

¹ Geometric mean values are based on N = 4.

² Multiple mean comparison based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

Indicates depth of amendment incorporation.

Soluble Arsenic levels
STARS Site 21 (Ramsay Flats)



Soluble Copper levels
STARS Site 21 (Ramsay Flats)

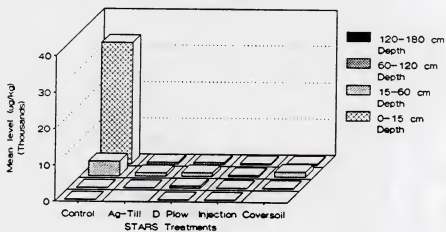


Figure 4.2.8. Mean water soluble As (top) and Cu (bottom) levels at STARS Site 21- Ramsay Flats arranged by profile depth and treatment.

Table 4.2.5. Mean concentrations¹ of water soluble metals ($\mu\text{g/kg}$) in amended wastes/soils at STARS Site 27.

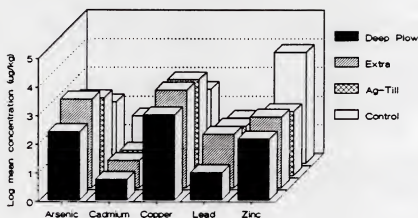
Amended Waste/ Soil Depth (cm)	Treatment			
	Control	Ag Till	Deep Plow	Extra
--- Arsenic ---				
0-15	141 A ²	525 AB	240 AB	1271 B
15-60	10 A	28 A	25 A	18 A
60-120	21	24	20	24
120-180	--	--	--	--
--- Cadmium ---				
0-15	46 A	7.5 A	5.2 A	8.9 A
15-60	3.8 A	4.5 A	8.4 A	2.2 A
60-120	2.4	1.9	9.5	116
120-180	--	--	--	--
--- Copper ---				
0-15	385 A	2249 A	925 A	2564 A
15-60	26 A	97 AB	201 B	40 A
60-120	33	48	200	5272
120-180	--	--	--	--
--- Lead ---				
0-15	38 A	52 A	9.0 A	72 A
15-60	1.4 A	2.4 A	3.2 A	1.2 A
60-120	2.8	2.4	8.1	261
120-180	--	--	--	--
--- Zinc ---				
0-15	7499 B	173 A	131 A	292 A
15-60	2506 A	537 A	693 A	269 A
60-120	219	96	945	19588
120-180	--	--	--	--

¹ Geometric mean values are based on N = 1, 2, 3 or 4 observations.

² Multiple mean comparison based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

Indicates depth of amendment incorporation.

STARS Site 27 (Fairmont, 0-15 cm)



STARS Site 33 (Opportunity, 0-15 cm)

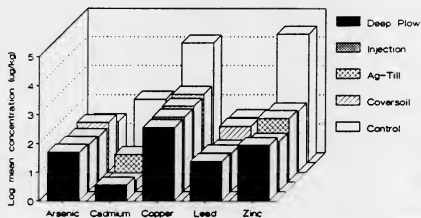


Figure 4.2.9. Mean water soluble metal levels (0-15 cm) for STARS Site 27 (Fairmont) and STARS Site 33 (Opportunity).

it was significant only at the extra treatment. Concentrations of Zn were significantly reduced in the 0-15 cm layer for all STARS treatments compared to the untreated materials in the control plot. These relationships are graphically presented in Figure 4.2.9, top. Below the zone of incorporation for any treatment there were no significant differences among the STARS treatments or between the control and any treatment for the soluble metals.

Water soluble metal and As mean concentration values for materials at Site 33 (Opportunity) are exhibited in Table 4.2.6. There was insufficient data to perform ANOVA for the levels found in the 0-15 cm layer. However, the general trend of reduction in mean concentrations of Cd, Cu and Zn were demonstrated. These relationships are displayed in Figure 4.2.9 (bottom). Changes in the mean levels of As or Pb were not apparent.

At the 15-60 cm depth materials amended using the deep plow revealed significantly lower Cd, Cu and Zn means compared to the materials in the control and injection plots. At the deepest incorporation zone, no concentration differences were found among the treatments.

Data for water soluble metals supporting tables and figures in this section are presented in Volume III, Appendix B1.

4.2.2 Pore Water Chemistry

Use of suction lysimeters allows collection of a small sample of soil solution from a relatively undisturbed soil profile. This technique provides the most unbiased method for investigating the chemical make-up of the soil pore water. Seasonal or other temporal changes in soil solution chemistry can also be investigated by repeatedly sampling the same lysimeter. A drawback of suction lysimeters is that they obtain a sample representative of only a small soil volume, hence unless they are replicated, treatment comparisons will be masked if the soil solution chemistry is spatially variable.

Samples are collected by creating a partial vacuum within the suction lysimeter and allowing enough time (approximately 24 hours) for pore water held under suction in the soil to be drawn through the lysimeter's ceramic tip. When the soil is dry either due to surface evaporation or plant water use, the partial vacuum created in the lysimeter is not sufficient to create a suction gradient that will draw pore water into the ceramic cup. As a consequence samples cannot be collected from dry soil. However, when the soil system is too dry to allow sample collection, it is also too dry for convective water movement to occur.

Lysimeters were installed at selected depths in one replicate of each treatment at each site. There were no injection or coversoil plots at site 27. An extra plot consisting of a high phosphate treatment was installed and monitored at site 27. Depths of lysimeter installation were 40, 90 and 150 cm on the control, deep plow and injection plots and 40 and

90 cm on the agricultural tillage (Ag-till), cover soil and extra plots. For details of plot and monitoring installation, refer to section 3.0.

Pore water samples were collected from lysimeter at each site during 1990, 1991 and 1992. Samples were analyzed for pH, EC and Eh in the field during all sampling trips and for major cations, anions and metals once during each sampling season. These results are presented in Volume II of the Appendix, with the corresponding field parameters. Samples were collected at site 2 (Manganese Stockpile) only during 1990. Only one sample was obtained throughout the study from the extra plot at site 27, thus no discussion of this treatment will take place.

Three types of figures are presented in the following discussion including graphs showing temporal changes in field-measured pH and EC by treatment at each site. Separate graphs for each depth monitored (40, 90 and 150 cm) are presented. These field parameters were measured several times per year for each lysimeter. A series of figures also portray relative differences in pore water chemistry for the single set of pore water samples collected in each year from 1990 to 1992. A series of pie charts corresponds to the sample characteristics taken from each treatment, and each monitored depth. Separate sets of pie charts are prepared for each site and each year. The area of each pie is proportional to the EC of the sample. Each pie shows the relative percentage of three groups of metals including alkali metals (Ca, Mg, Na, K); low toxicity metals (Al, Fe, Mn); and higher toxicity metals (Cd, Cu, Pb, Zn). All percentages represent the molar fraction of each group of metals. The EC value correlates with the total molar sum of cations. The metals were broken into the three groups based on their relative toxicity characteristics (eg. $Cd > Fe > Ca$) and concentrations at which they occur in natural soils (eg. $Ca > Fe > Cd$). Samples were not analyzed for As in all years, thus As was not included in the figures. When samples were analyzed for As, it was less than 0.2 % of the total cation concentration in all samples with the exception of 2 samples from the 90 cm depth on the control site at site 7 in 1991 and 1992 (1.4 and 1.8 %, respectively). Finally a series of horizontal bar charts were prepared to portray the absolute quantity of specific metals at each monitored depth for two selected treatments; the control and deep plow. Data for all three years are included in each set of graphs.

Table 4.2.6. Mean concentrations¹ of water soluble metals ($\mu\text{g/kg}$) in amended wastes/soils at STARS Site 33.

Amended Waste/ Soil Depth (cm)	Treatment				
	Control	Ag Till	Deep Plow	Injection	Coversoil
--- Arsenic ---					
0-15	24	35	48	41	49
15-60	12 A ²	18 A	28 A	10 A	13 A
60-120	86 A	86 A	33 A	22 A	9.9 A
120-180	25	—	42	30	—
--- Cadmium ---					
0-15	142	8.3	3.5	2.3	2.9
15-60	80 C	17 BC	3.1 A	18 BC	8.5 AB
60-120	2.4 A	1.4 A	2.2 A	5.5 A	10 A
120-180	0.8	—	1.1	1.0	—
--- Copper ---					
0-15	12459	366	336	355	460
15-60	2366 B	367 A	113 A	316 A	157 A
60-120	53 A	41 A	91 A	64 A	376 A
120-180	26	—	47	25	—
--- Lead ---					
0-15	34	14	23	21	36
15-60	19 B	7.0 AB	2.2 A	9.6 B	8.2 B
60-120	1.8 A	0.8 A	1.7 A	1.0 A	3.6 A
120-180	0.6	—	0.5	1.3	—
--- Zinc ---					
0-15	26561	149	84	53	122
15-60	10116 C	1094 BC	76 A	1409 BC	619 AB
60-120	51 A	37 A	58 A	232 A	631 A
120-180	26	—	28	75	—

¹ Geometric mean values are based on N = 4.

² Multiple mean comparison based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

Indicates depth of amendment incorporation.

Addition of lime (CaCO_3 and $\text{Ca}(\text{OH})_2$) generally increased the pH of upper soil layers and as a consequence decreased the soluble levels of metals. Most lysimeters (with the exception of the deep plow and the injection plot) were placed below the zone of lime incorporation. As a result, notable changes in pore water chemistry would have to be attributed to the effects of the downward leaching of alkalinity from the surficially incorporated amendments. Hence the suction lysimeters were used primarily to assess the indirect effect of amendment application in the upper 10 to 30 cm (75 to 100 cm in the deep plow and injection plot) on the chemistry of deeper soil horizons. By contrast, the replicated soil sampling and analysis described in section 4.2.1 also measured chemical changes within the zone of amendment incorporation for each treatment. Pore water sampling results are interpreted in the following section in two different ways. First the temporal changes in solution chemistry and the differences in pore water between treatments is characterized. In the later half of this subsection, the generalized response of each metal in pore water to the primary effect of the lime treatment (*eg.* the increase in pH) is described.

In general, many of the deep plow and some of the lime slurry injection plots exhibited an increase in the percentage of alkaline earth cations (Ca, Mg, Na, K) and a decrease in trace metals (both Al, Fe, Mn and Cd, Cu, Pb and Zn) within the zones of incorporation. Because of the spatial variability of the tailings depositional environment and chemical characteristics and the fact that lysimeter instrumentation was not replicated, it is difficult in some cases to attribute differences in pore water chemistry between plots to a particular treatment. This is especially true when the lysimeter is installed in native soil located beneath the tailings, or below the depth of amendment incorporation. In some cases the control plots exhibit the lowest levels of metals in pore water. Although there were significant changes in the chemistry of a given lysimeter through time, the relative percentage of metals groups remains predominantly unchanged, suggesting that the treatments had an immediate and lasting effect on vadose zone chemistry.

Manganese Stockpile: Pore water data for site 2 is available only for 1990. Marked differences in pore water pH and EC (Figure 4.2.10) were noted especially at the 40 cm depth. Tailings at the Manganese Stockpile extend to 120 cm so the 40 and 90 cm lysimeters were in tailings. Only the deep plow and injection plots had lysimeters placed within the potential zone of incorporation.

The injection, cover-soil and ag-till plots had the lowest 40 cm pH levels (below 4.0) while the deep plow plot had the highest pH (5 to 6). The control and extra plot had the highest EC (greater than 14 mmhos/cm). Solution pH levels varied more than 1.5 units between sampling dates but no clear temporal trend emerged for all plots. Changes in pH and EC could be caused by desiccation leading to an increase in dissolved concentrations (EC) and to precipitation of various solids which in turn could cause pH shifts. In addition, increases in biological activity as the soil warmed could increase CO_2 evolution leading to a decline in pH with little change in EC. Finally, recharge events could dissolve ferric sulfate salts (increased EC) leading to subsequent hydrolysis reactions and a steep decline in pH. The magnitude of the pH variation was much greater than expected and was most

pronounced at 40 cm depth and in unamended layers. The cover-soil treatment and the deep plow plot exhibited the least variation in pH and EC temporally.

Comparison of metal levels in pore water at site 2 (Figure 4.2.11) indicated that deep plowing resulted in a sizeable decrease in the percentage of Al, Fe and Mn and an increase in alkaline earth metals relative to the other treatments. A slight increase in pH (approximately 0.5) was also noted relative to the control at both the 40 and 90 cm depth. Injection also reduced the percentage of Al, Fe, Mn and Cd, Cu, Pb and Zn relative to the control, however the low pH and high EC of the 40 cm sample in the injection plot indicates that pH was not adequately controlled by slurry lime injection. Comparison of the pore water chemistry in the control versus the ag-till, and cover-soil plot indicates that surficial application of lime did not increase pH at greater depths in the soil profile. Only deep plowing may have had a beneficial effect on pH at 40 to 90 cm although the pH improvement was not as great as that expected at the design lime application rate. Inadequate mixing of lime with soil material at depth was the likely cause of poor pH control. Photographs illustrating the degree of amendment mixing (Section 4.2.4) clearly show distinct zones of low and high pH at depth. Roughly about 30 to 70% of the 2 to 4 foot depth of soil exhibited an increase in pH.

The absolute concentrations of metals in pore water were compared for the control and deep plow plot (Figure 4.2.12). Conspicuous differences in metal levels (copper, zinc, and arsenic) did not exist with the possible exception of the deep plow 90 cm depth which had higher pH and dramatically lower zinc, iron and manganese concentrations than the control.

Pore water pH levels varied from 3 to 5 at 40 cm and were between 5 and 6 at 90 and 150 cm in depth. Although salinity levels were widely variable, the highest levels which were well above the tolerance of most plants were noted at 40 cm (above 20 mmhos/cm). Levels of zinc, manganese, and to a lesser degree copper were highly elevated at all depths monitored, even below the deepest extent of tailings and essentially at or below the groundwater interface.

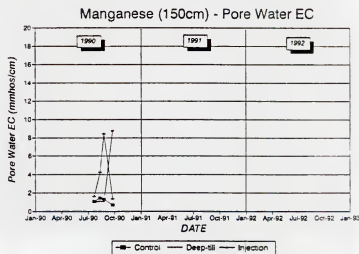
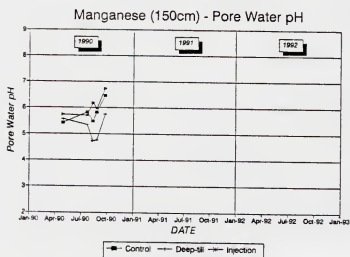
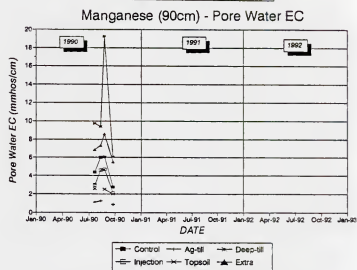
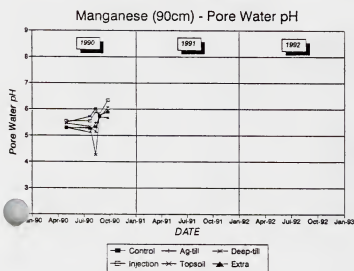
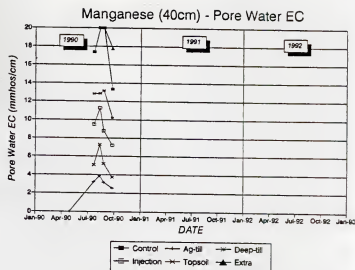
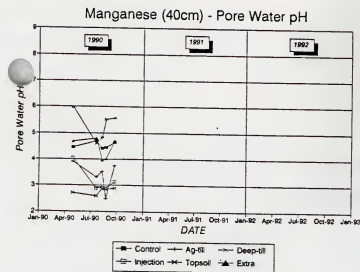
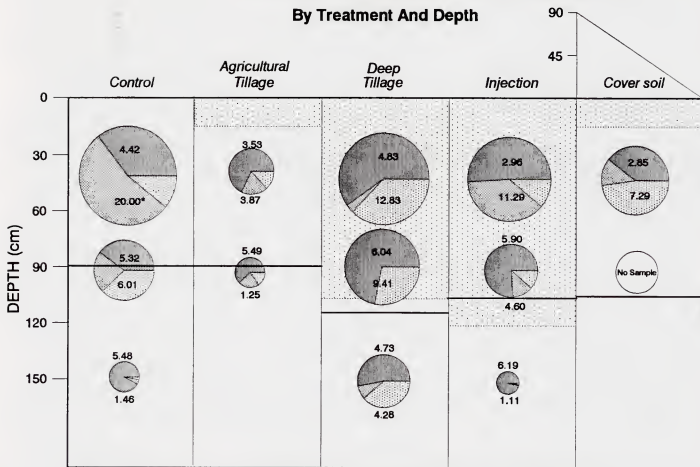


Figure 4.2.10.

Temporal trends in pore water pH and EC at the Manganese Stockpile in 1990.

SITE 02 - 1990 PORE WATER CHEMISTRY

By Treatment And Depth



* Note: SC value for 40cm control plot lysimeter=20.00 but pie size is equivalent to SC=15 for scaling purposes.

Note: Cover soil depth not to scale.

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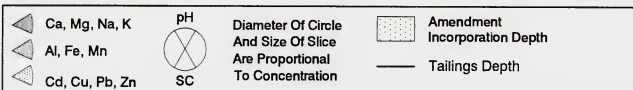
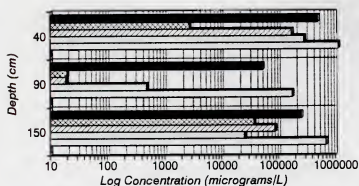


Figure 4.2.11. Pore water 1990 metal chemistry at site 02 (Manganese Stockpile).

MANGANESE DEEP-TILL PLOT 1990
Pore Water Cation Concentration



MANGANESE CONTROL PLOT 1990
Pore Water Cation Concentration

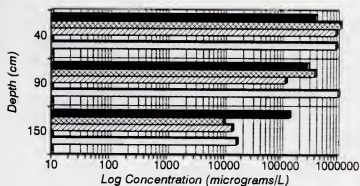


Figure 4.2.12. Concentrations of key ions in pore water at various depths in the control and deep plow plots in 1990.

Rocker: Tailings range from 50 to 100 cm in thickness at Rocker where they are commonly stratified with cleaner sediments. Groundwater occurs at 120 to 140 cm in depth so the deepest lysimeter sample represents the shallow alluvial groundwater. Pore water pH levels vary from 4 to 7 at 40 cm, 2 to 6.5 at 90 cm, and are more uniform at 150 cm at around 6.5. Salinity ranges from 1 to 6 but averages close to 2 at all depths.

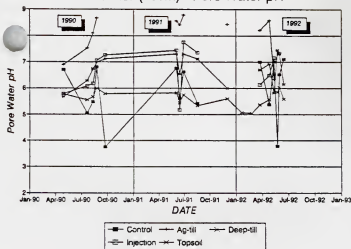
Large seasonal fluctuation in pH was more pronounced at Rocker, particularly at 40 cm, than at any other site (Figure 4.2.13). Wide variation in pH, from less than 4 to over 7 in the control, may have been due to the coarse texture and weak buffering ability of the soil at Rocker. Salinity levels were much lower at Rocker than at the Manganese Stockpile. A general trend for increasing pH during the growing season may have been due to gradual drying of the soil profile allowing for more rapid exchange of CO₂ evolved by roots and microorganisms. The upward pH trend appeared to be interrupted by sharp, short-duration declines in pH which may have occurred in response to recharge events (Figure 4.2.14)

Major constituents in pore water sampled in 1990 (Figure 4.2.15) indicate that the chemistry of most treatments, including the control, is dominated by alkaline earth cations. Other metals make up a small percentage of the control and injection samples collected from the 90 and 150 cm depths. Deep plowing and injection resulted in an increase in pH of approximately one unit relative to the other treatments, especially at the 40 cm depth. The sample collected from the Ag-till plot at 90 cm consistently had a much lower pH and greater EC and percentages (and concentrations) of metals relative to all the other samples and likely represents a localized highly acid "hot spot".

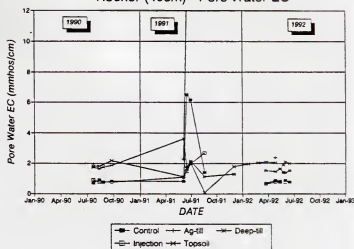
Pore water results from 1991 and 1992 (Figures 4.2.16 and 4.2.17) were nearly identical with respect to overall metal percentages for all samples. Samples from the deep plow treatment did show a trend of slightly increasing pH through time and most treatments, including the control, showed increases in pH from 1990 to 1991. The relatively low EC values for all samples at site 7 relative to the other sites is probably due to the coarse nature of the tailings and native materials as a result of their location adjacent to a high-energy stream reach. Similarly, the groundwater table may have affected the pore water results at the 150 cm depth.

Metal levels at 40 cm measured in 1990, 1991 and 1992 are somewhat higher in the deep plow than in the control, while the opposite was true at 90 and 150 cm (Figure 4.2.18). Metals levels at Rocker as at the Manganese Stockpile were highly elevated at all depths including at 150 cm which was below the deepest extent of tailings and was in the upper portion of the groundwater system. Metals have clearly been leached from tailings into the groundwater system at Rocker. Due to the variability of the soils monitored and the lack of replication it is difficult to ascribe differences in pore water chemistry between plots to treatment effects. The only consistent difference in pore water make-up between the control and deep plow is the lower iron and higher arsenic in the deep plow. Copper and to a lesser degree manganese and zinc appear to be lower in the deep plow plot.

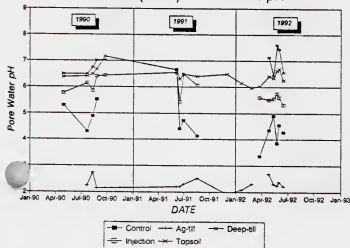
SILVER BOW CREEK
Rocker (40cm) - Pore Water pH



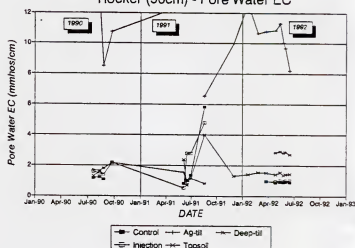
SILVER BOW CREEK
Rocker (40cm) - Pore Water EC



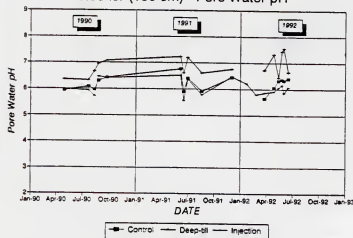
Rocker (90cm) - Pore Water pH



Rocker (90cm) - Pore Water EC



Rocker (150 cm) - Pore Water pH



Rocker (150cm) - Pore Water EC

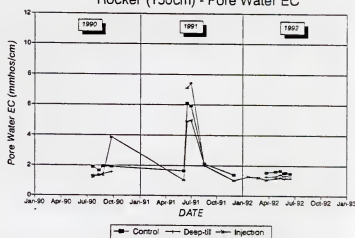


Figure 4.2.13. Temporal trends in pore water pH and EC at Rocker in 1990.

RAMSAY FLATS CLIMATE STATION

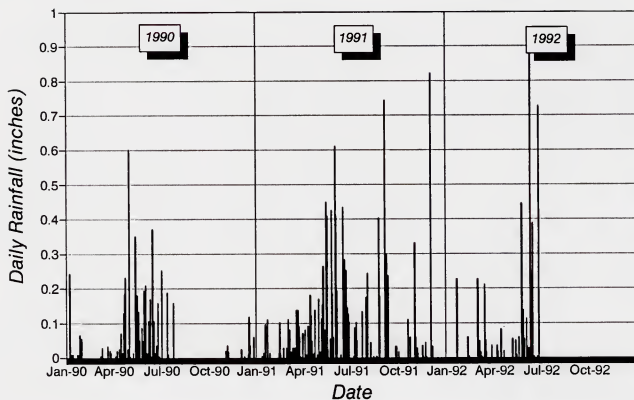
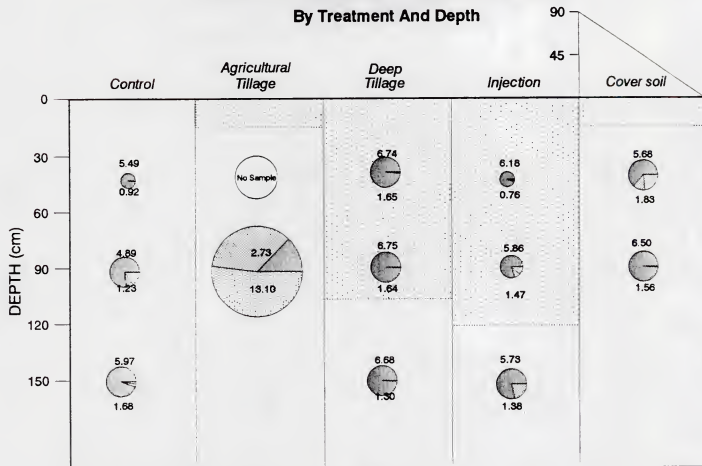


Figure 4.2.14. Daily rainfall quantity measured in 1990, 1991, and 1992 at Ramsay Flats. High rainfall amounts generally correspond to sharp, short-duration reductions in pH.

SITE 07 - 1990 PORE WATER CHEMISTRY

By Treatment And Depth



* Note: Tailings mixed with natural alluvium and not discretely observable.

Note: Cover soil depth not to scale.

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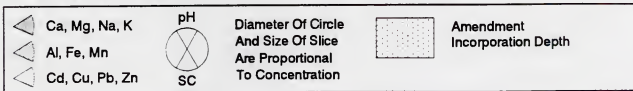
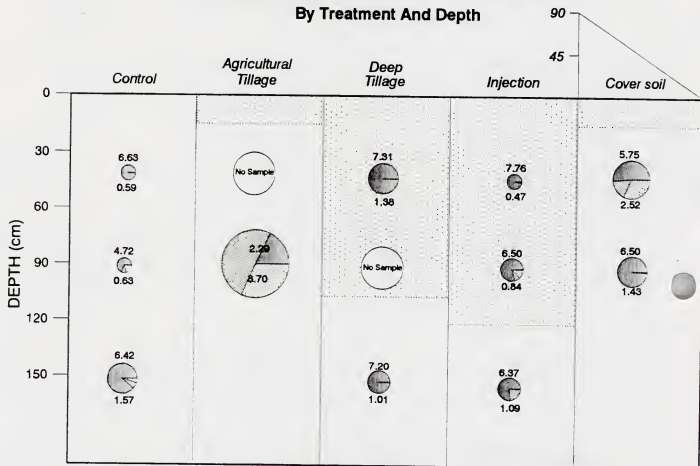


Figure 4.2.15. Pore water 1990 metal chemistry at site 07 (Rocker).

SITE 07 - 1991 PORE WATER CHEMISTRY **By Treatment And Depth**



* Note: Tailings mixed with natural alluvium and not discreetly observable.

Note: Cover soil depth not to scale.

LEGEND

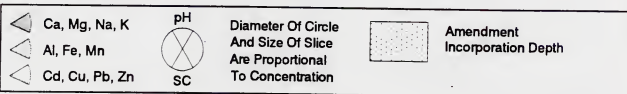
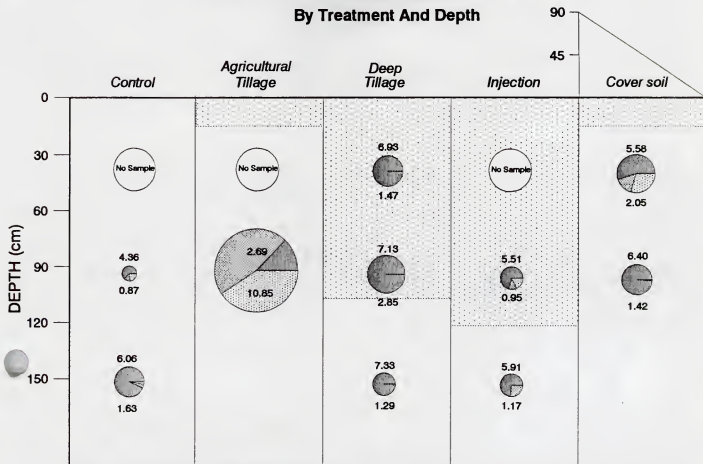


Figure 4.2.16. Pore water 1991 metal chemistry at site 07 (Rocker).

SITE 07 - 1992 PORE WATER CHEMISTRY

By Treatment And Depth



* Note: Tailings mixed with natural alluvium and not discretely observable.

Note: Cover soil depth not to scale.

LEGEND

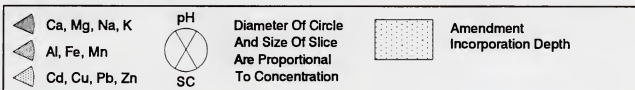
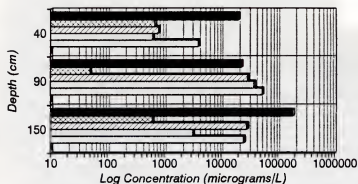


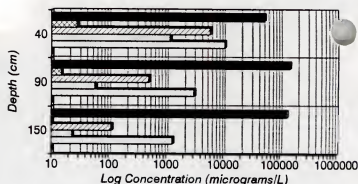
Figure 4.2.17.

Pore water 1992 metal chemistry at site 07 (Rocker).

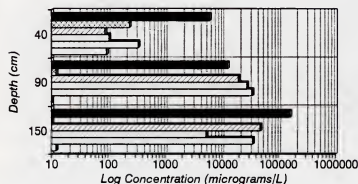
ROCKER CONTROL PLOT 1990
Pore Water Cation Concentration



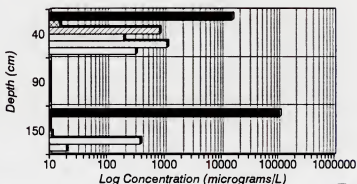
ROCKER DEEP-TILL PLOT 1990
Pore Water Cation Concentration



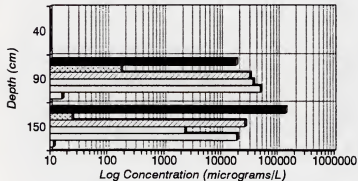
1991



1991



1992



1992

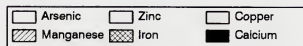
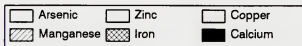
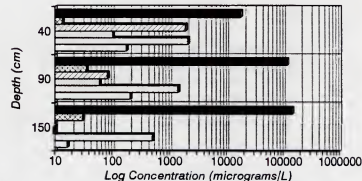


Figure 4.2.18.

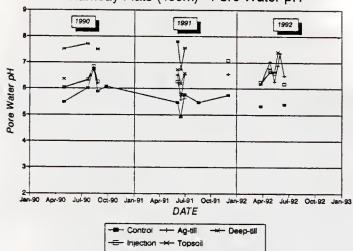
Concentrations of key ions in pore water at various depths in the control and deep plow plots from 1990 through 1992.

Ramsay Flats: Tailings at Ramsay Flats are much finer-textured at Ramsay and are thinner (40 to 50 cm) than at either Rocker or at the Manganese Stockpile. Depth to groundwater was generally greater than 200 cm. The pH of pore water averaged 5.5 to 6.5 near the base of the tailings at 40 cm, and ranged from 6.5 to 8 at 90 and 150 cm. Seasonal variations in pH and EC (Figure 4.2.19) although present were much less prevalent than at Rocker or the Manganese Stockpile. The control plot had the lowest pH at all depths while the deep plow exhibited the highest pH levels at 40 and 150 cm. The injection plot had the highest pH at 90 cm.

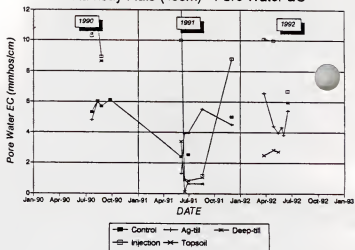
Treatment effects at Ramsay Flats were similar to those observed at Rocker, namely that deep plowing and injection resulted in an increase in the percentage of alkaline earth metals at the expense of Al, Fe, Mn and Cd, Cu, Pb and Zn within the zone of incorporation (Figures 4.2.20 through 4.2.22). The Ag-till and cover soil treatments yielded similar results at the 40 cm depth. The decrease in Al, Fe, Mn and Cd, Cu, Pb and Zn percentages in control samples (and all treatment samples) at the 90 and 150 cm depths is likely due to the presence of natural soil at this depth. Deep plowing and injection resulted in an increase in EC relative to the control and other treatments. Values of pH were fairly high for all treatments, including the control, and ranged between approximately 6 and 7.5, with most amended zones showing a small increase in pH relative to the control. Treatment response remained consistent through time.

Figure 4.2.23 compares metal levels between the control and deep plow plots at all depths in 1990 through 1992. Highly elevated metal levels are mostly confined to the 40 and 90 cm depth in the control plot. The deep plow had substantially lower iron, copper, zinc, iron and manganese than the control at 40 cm, but also had higher arsenic levels. The buried soil system at Ramsay has performed as a fairly effective geochemical barrier preventing widespread acidification and well-defined metal contamination in buried natural soils. Factors tending to reduce the leaching of metals at Ramsay when compared to Rocker may include finer soil textures, higher organic matter levels, greater thickness of the unsaturated zone beneath the tailings and reduced quantities of percolation.

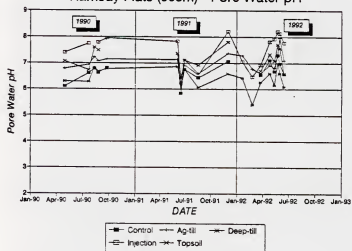
SILVER BOW CREEK
Ramsay Flats (40cm) - Pore Water pH



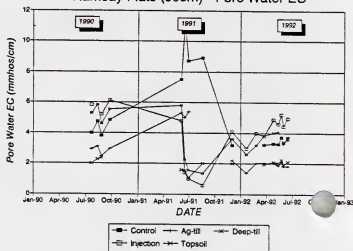
SILVER BOW CREEK
Ramsay Flats (40cm) - Pore Water EC



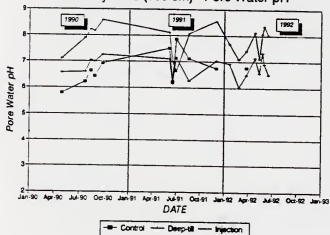
Hamsay Flats (90cm) - Pore Water pH



Ramsay Flats (90cm) - Pore Water EC



Ramsay Flats (150 cm) - Pore Water pH



Ramsay Flats (150cm) - Pore Water EC

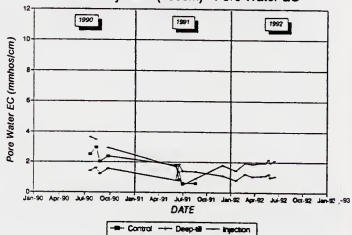
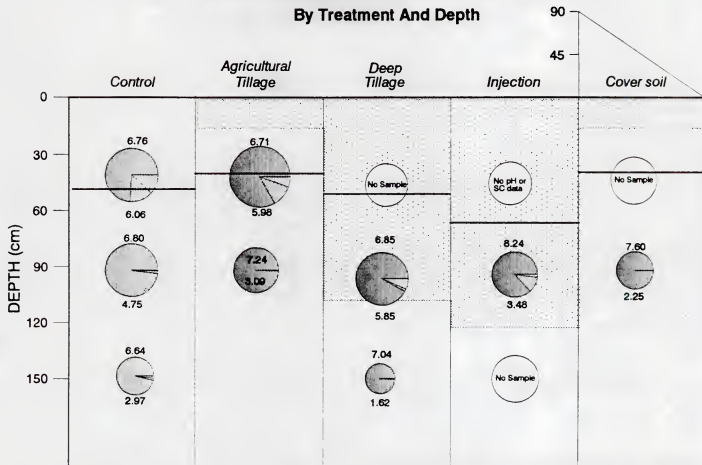


Figure 4.2.19. Temporal trends in pore water pH and EC at Ramsay in 1990.

SITE 21 - 1990 PORE WATER CHEMISTRY

By Treatment And Depth



* Note: Cover soil depth not to scale.

LEGEND

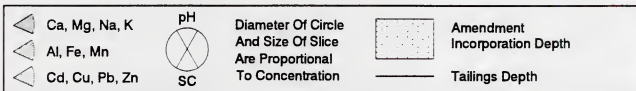
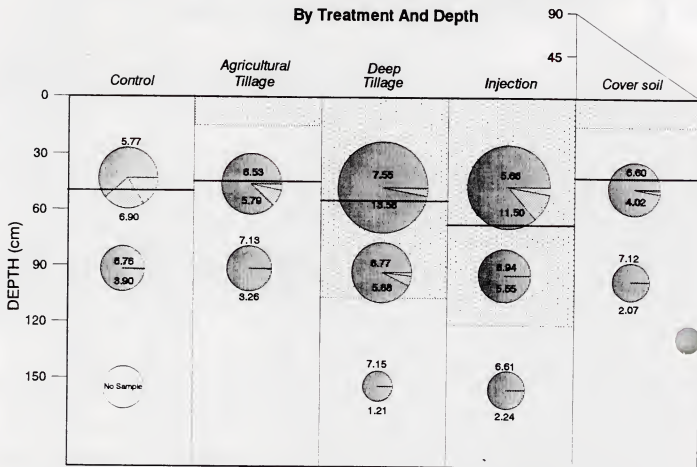


Figure 4.2.20. Pore water 1990 metal chemistry at site 21 (Ramsay Flats).

SITE 21 - 1991 PORE WATER CHEMISTRY **By Treatment And Depth**



* Note: Cover soil depth not to scale.

LEGEND

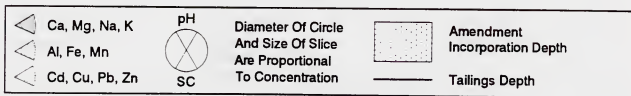
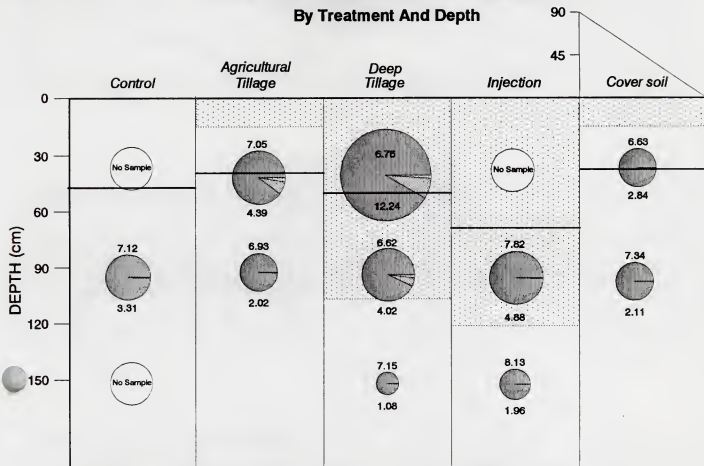


Figure 4.2.21. Pore water 1991 metal chemistry at site 21 (Ramsay Flats).

SITE 21 - 1992 PORE WATER CHEMISTRY

By Treatment And Depth



* Note: Cover soil depth not to scale.

LEGEND

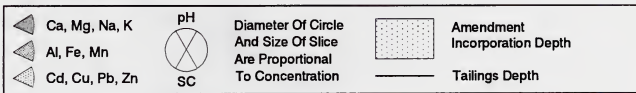
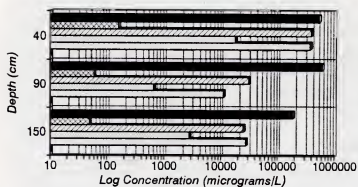
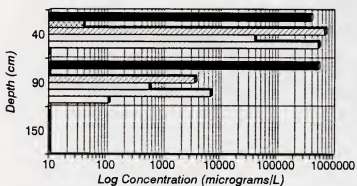


Figure 4.2.22. Pore water 1992 metal chemistry at site 21 (Ramsay Flats).

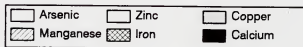
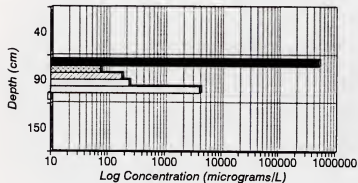
RAMSAY CONTROL PLOT 1990
Pore Water Cation Concentration



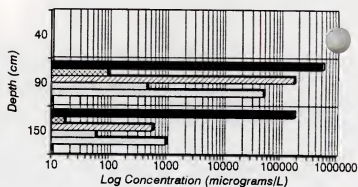
1991



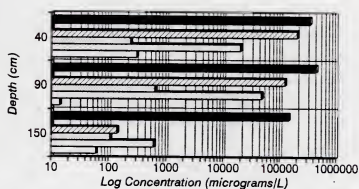
1992



RAMSAY DEEP-TILL PLOT 1990
Pore Water Cation Concentration



1991



1992

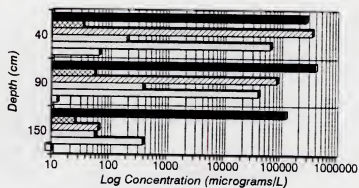


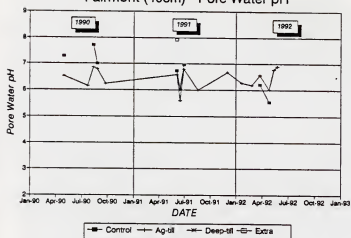
Figure 4.2.23. Concentrations of key ions in pore water at various depths in the control and deep plow plots from 1990 through 1992.

Fairmont: The Fairmont site is in an upland setting with no observed groundwater influence or capillary fringe. A thin layer of 10 to 20 cm of tailings was spread over natural soils by inadvertent historic irrigation with contaminated water. All pH values were between 6 and 8 at all depths (Figure 4.2.24). The control had the highest EC at 40 and 90 cm (6 to 10 mmhos/cm). Pore water pH levels were not as variable as at Rocker, however EC levels tended to change abruptly during the growing season.

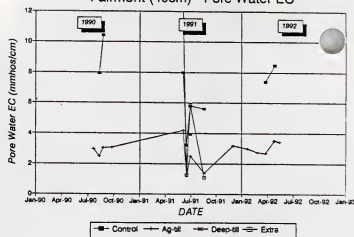
The injection and cover soil treatments were not employed at site 27. No samples were obtained at any time from any depth for the deep plow treatment (Figures 4.2.25 through 4.2.27). The dry soil condition in the deep plow plots may have been a result of increased evapotranspiration in response to vegetation growth. At depths for which data is available, the Ag-till treatment had little effect on the pore water chemistry versus the control. Values for pH are very similar at 40 cm and approximately one unit higher at 90 cm. It is difficult, however, to determine if this increase is a function of treatment or spatial variability, especially since a pH value of 7.51 was obtained from the control sample at 150 cm in 1992. Conductivity (SC) values are lower in the Ag-till treatment samples.

Metal levels in pore water were generally lower at all depths at Fairmont than at other sites (Figure 4.2.28). No clearcut effect on metal levels at any depth could be attributed to deep tillage at Fairmont.

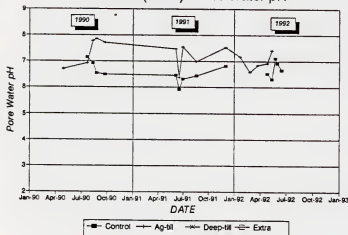
SILVER BOW CREEK
Fairmont (40cm) - Pore Water pH



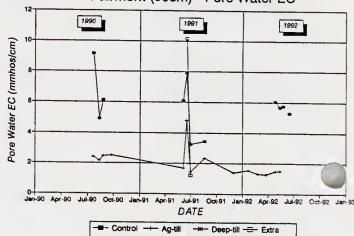
SILVER BOW CREEK
Fairmont (40cm) - Pore Water EC



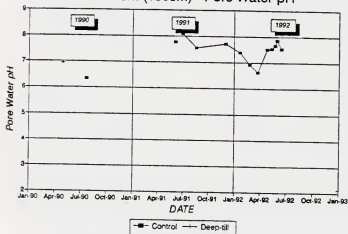
Fairmont (90cm) - Pore Water pH



Fairmont (90cm) - Pore Water EC



Fairmont (150cm) - Pore Water pH



Fairmont (150cm) - Pore Water EC

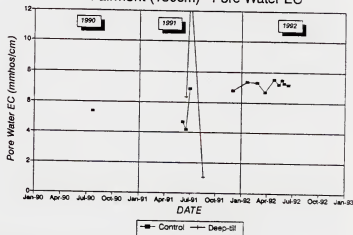
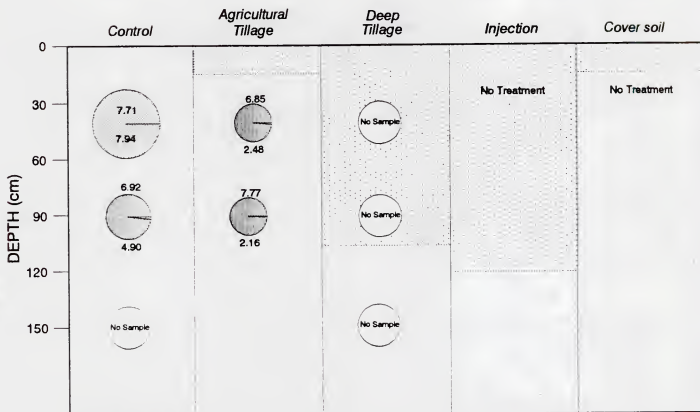


Figure 4.2.24. Temporal trends in pore water pH and EC at Fairmont.

SITE 27 - 1990 PORE WATER CHEMISTRY **By Treatment And Depth**



* Note: No tailings discernable.

LEGEND

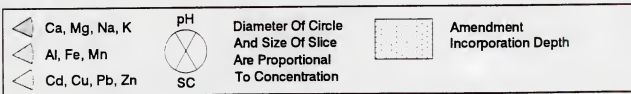
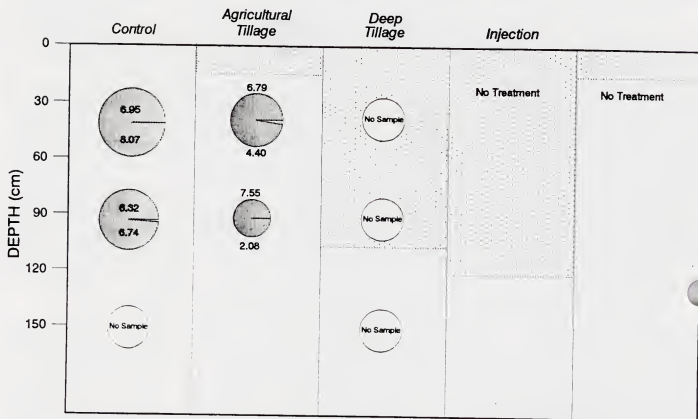


Figure 4.2.25. Pore water 1990 metal chemistry at site 27 (Fairmont).

SITE 27 - 1991 PORE WATER CHEMISTRY **By Treatment And Depth**



* Note: No tailings discernable.

LEGEND

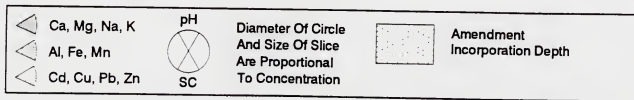
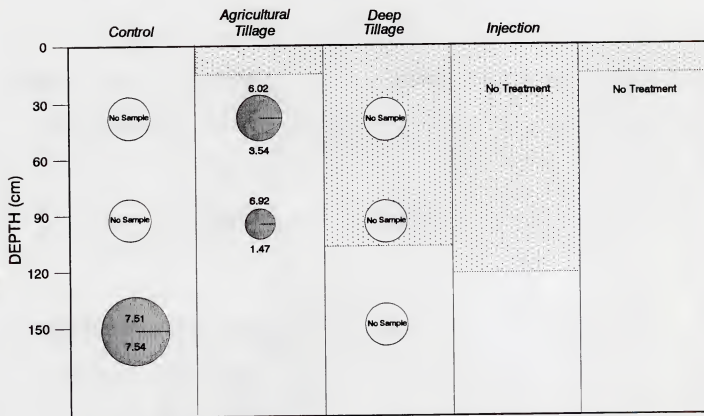


Figure 4.2.26. Pore water 1991 metal chemistry at site 27 (Fairmont).

SITE 27 - 1992 PORE WATER CHEMISTRY **By Treatment And Depth**



* Note: No tailings discernable.

LEGEND

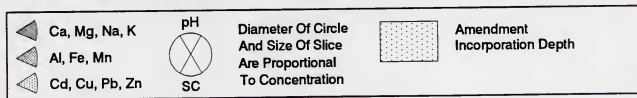
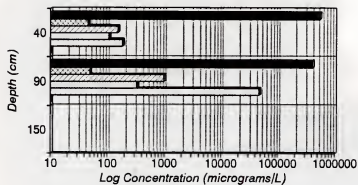
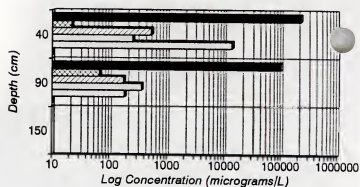


Figure 4.2.27. Pore water 1992 metal chemistry at site 27 (Fairmont).

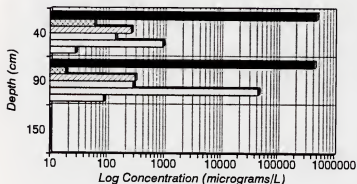
FAIRMONT CONTROL PLOT 1990
Pore Water Cation Concentration



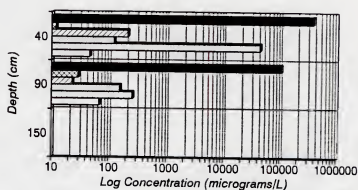
FAIRMONT AG-TILL PLOT 1990
Pore Water Cation Concentration



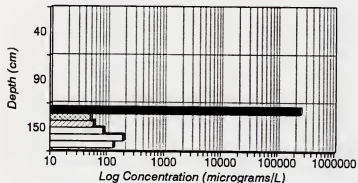
1991



1991



1992



1992

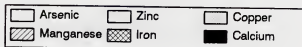
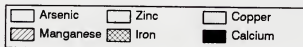
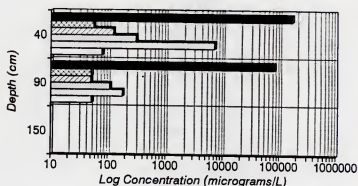


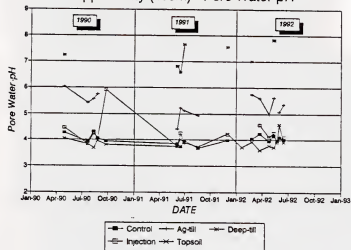
Figure 4.2.28. Concentrations of key ions in pore water at various depths in the control and deep plow plots from 1990 through 1992.

Opportunity: Approximately 20 to 40 cm of discontinuous sandy tailings were found at the Opportunity site which had groundwater at 150 to 200 cm in depth. Pore water pH levels at 40 cm were near 4 except for the ag-till and deep plow plots which had pH levels of 6 and 7.5 respectively. Solution pH and EC levels were relatively stable compared to other sites (Figure 4.2.29) with the exception of a late June sample in 1991 which had noticeably lower pH and higher EC perhaps due to an isolated percolation event in response to a thunderstorm.

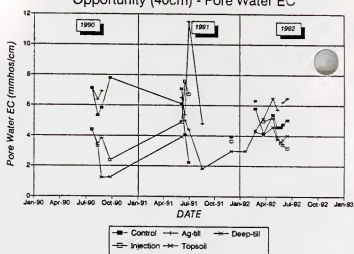
Samples from 90 and 150 cm for all treatments in all years (Figures 4.2.30 through 4.1.32) are dominated by alkaline earth cations, due to the presence of relatively uncontaminated native soil material at these depths. The Ag-till treatment showed an increase in alkaline earth cations and associated decrease in Al, Fe, Mn and Cd, Cu, Pb and Zn relative to the control and other treatments at the 40 cm depth during 1990. No sample was obtained from the deep plow plot in 1990 and 1992 at the 40 cm depth. However, a sample obtained in 1991 showed a considerable reduction in EC and very large (2 to 4 units) increases in pH relative to all other treatments. Percentages of Al, Fe, Mn, and Cd, Cu, Pb and Zn were reduced to < .02 % of total cations compared to the control (32 %). All other treatments resulted in a small decrease in SC relative to the control. Overall, pH increased slightly and EC declined with depth in all treatments.

Figure 4.2.33 indicates that highly elevated metal levels were mostly confined to the 40 cm depth in the control plot at Opportunity. Leaching of metals from the tailings into underlying soil (hence into groundwater) was less pronounced at Opportunity than at any other site. Factors tending to reduce the downward flux of metals may include finer soil textures, higher organic matter levels, greater thickness of the unsaturated zone beneath the tailings and reduced quantities of percolation. Deep plowing clearly reduced the quantity of iron, copper, zinc and manganese and may have slightly increased soluble arsenic levels at 40 cm at Opportunity. No increase in soluble arsenic was noted at depth, however.

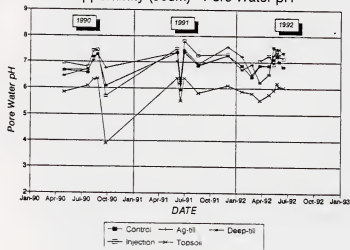
SILVER BOW CREEK
Opportunity (40cm) - Pore Water pH



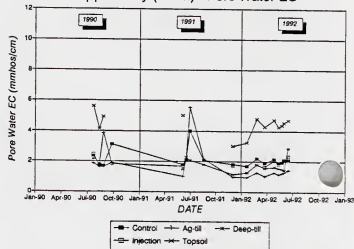
SILVER BOW CREEK
Opportunity (40cm) - Pore Water EC



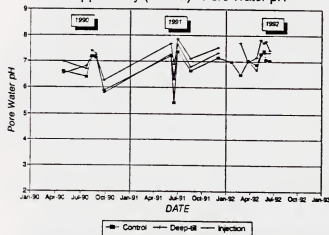
Opportunity (90cm) - Pore Water pH



Opportunity (90cm) - Pore Water EC



Opportunity (150cm) - Pore Water pH



Opportunity (150cm) - Pore Water EC

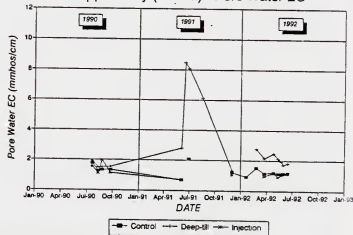
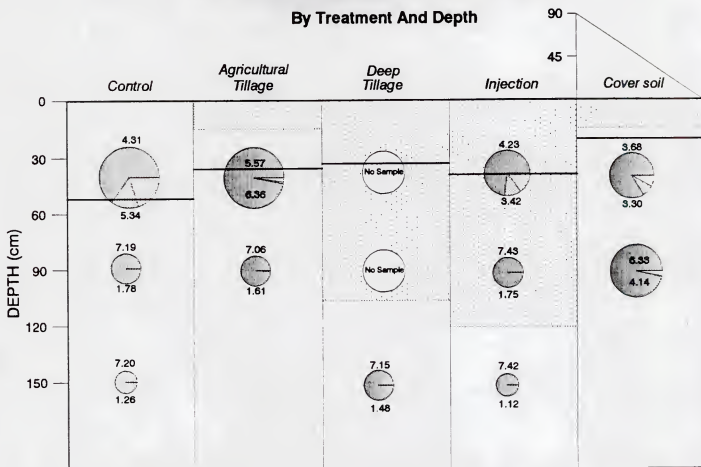


Figure 4.2.29. Temporal trends in pore water pH and EC at Opportunity.

SITE 33 - 1990 PORE WATER CHEMISTRY

By Treatment And Depth



* Note: Cover soil depth not to scale.

LEGEND

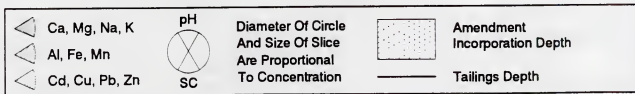
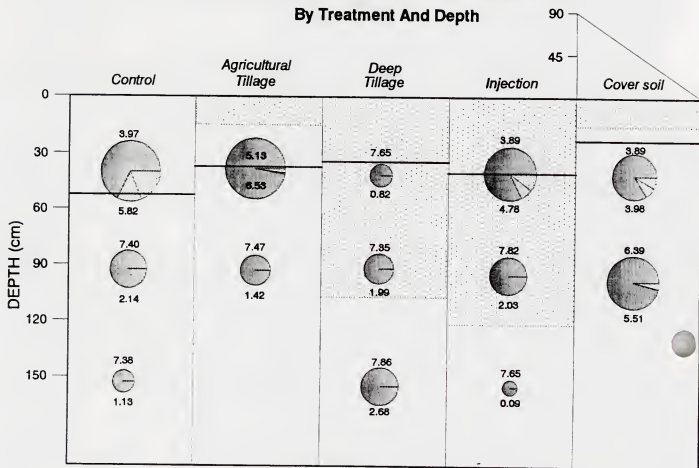


Figure 4.2.30. Pore water 1990 metal chemistry at site 33 (Opportunity).

SITE 33 - 1991 PORE WATER CHEMISTRY

By Treatment And Depth



* Note: Cover soil depth not to scale.

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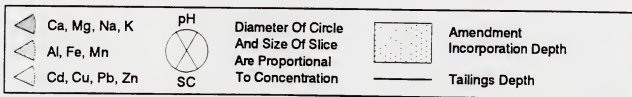
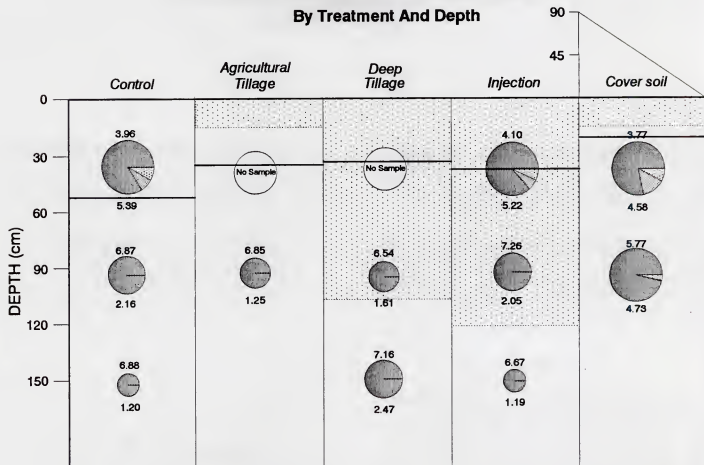


Figure 4.2.31. Pore water 1991 metal chemistry at site 33 (Opportunity).

SITE 33 - 1992 PORE WATER CHEMISTRY

By Treatment And Depth



* Note: Cover soil depth not to scale.

LEGEND

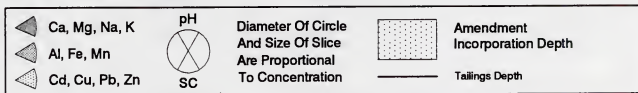
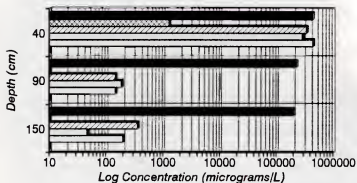
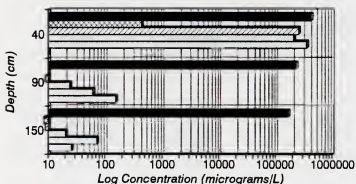


Figure 4.2.32. Pore water 1992 metal chemistry at site 33 (Opportunity).

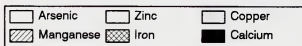
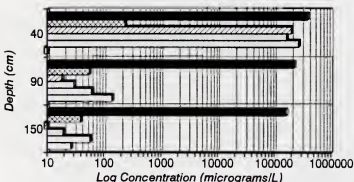
OPPORTUNITY CONTROL PLOT 1990
Pore Water Cation Concentration



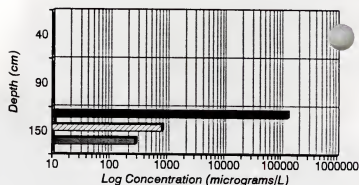
1991



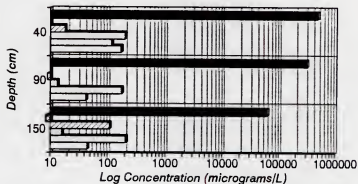
1992



OPPORTUNITY DEEP-TILL PLOT 1990
Pore Water Cation Concentration



1991



1992

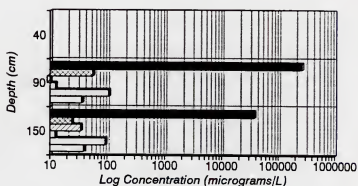


Figure 4.2.33. Concentrations of key ions in pore water at various depths in the control and deep plow plots from 1990 through 1992.

Summary of Pore Water Chemistry: Significant spatial and temporal variability characterized much of the pore water data, however several important conclusions can be made about the geochemistry of the control and amended plots at each site. While solution pH is an important variable which affects the solubility of most metal cations, the pH of the vadose zone can vary through time by two to three units potentially due to a number of factors. Increases in biological activity can increase dissolved CO_2 levels as the temperature warms which may lead to a decline in pH. Conversely plant growth may dry the soil leading to an increase in the gas exchange rate causing a decline in dissolved CO_2 . Sudden and extreme decreases in pore water pH with concurrent increases in EC were thought to be due to movement of pore water from upper soil layers where ferric sulfates and other salts dissolved leading to hydrolysis of iron and a decline in pH.

Most pore water samplers were installed below the depth of amendment incorporation. In general, no difference in pore water chemistry was observed between the control, ag-till and cover-soil treatments indicating that downward leaching of alkalinity was negligible. The deep-plow treatment often exhibited an increase in solution pH and a decrease in iron, copper, and cadmium compared to the control plot at both the 40 and 90 cm depth. Less often a decrease in manganese and zinc and an increase in arsenic were also noted. Slurry injection appeared to be less effective in controlling solution pH at most sites.

At the Manganese Stockpile site and at Rocker, tailings were roughly 100 cm thick while groundwater occurred at 100 to 130 cm in depth, hence there was less than 50 cm of vertical separation between tailings and groundwater. At both of these sites elevated levels of metals were detected at 150 cm, well below the tailings zone. Downward leaching had clearly caused localized groundwater degradation at these sites, at least in the surficial portion of the aquifer. Apparent contamination of sediment underlying the Rocker tailings may have been due to the coarse texture of the natural sediments which lent the material little buffering or metal ion adsorption capacity. In addition, it was extremely difficult to visually distinguish between contaminated and clean sediment at Rocker, hence the conclusion that natural sediments have been contaminated due to the downward flux of metals is subject to debate. At the Ramsay Flats and Opportunity site, tailings are 50 and 30 cm thick respectively while groundwater lies at least 100 cm below the tailings/natural soil interface. Natural soils at these sites have been an effective geochemical barrier confining metal contamination to the upper 30 to 50 cm of buried sediments.

General Trends in Pore Water Chemistry: Due to natural spatial variability of flood-deposited tailings and due to the variable amendment incorporation at depth, marked differences in pore water chemistry could not be attributed to the primary treatments. The pore water data collected can be useful in constructing some generalized conclusions regarding metal behavior in response to pH change. An understanding of the geochemical influence exerted by soil solution pH is especially important for two reasons. First, the primary effect of adding lime is to increase the system pH. Second, as contaminated pore water from the tailings zone percolates downward, the underlying soils tend to buffer the pH

of the solution. Hence, changes in pore water pH can either occur due to soil amendment of the tailings themselves or due to geochemical evolution of percolating seepage.

To examine the overall effects of amendment incorporation on improving vadose zone chemistry, solubility plots were constructed showing the calculated molarity of several specific constituents in pore water versus pH. Symbols used on each graph represent the following materials: T = tailings; S = natural soil; A = amended tailings; a = amended natural soil. In this way trends in metal concentrations can be either attributed to the presence of natural soil, amendment incorporation or both. All plots are for all samples which were analyzed for metals during 1992, the most recent data.

Overall, the addition of amendments resulted in an increase in pH between 1 and 2 units within the zone of incorporation at most of the sites. Differences in pH and SC between treatments and control natural soils below the zone of incorporation were minor and within the natural variability observed in control samples throughout the study. Also, the addition of amendment to natural soils did not notably decrease metal levels because the soils already were at near-neutral pH values.

Amendment incorporation decreased metals in pore water within the zone of incorporation, especially Cu and Al. Iron and Pb levels, which were relatively low in control samples, were also decreased to a lesser extent. Manganese, Cd and Zn behavior was similar, exhibiting a wide range of concentrations (several orders of magnitude) at near-neutral pH values. It was not until pH levels were increased to close to 8 that a decline in the levels of these metals was noted. In many cases, the higher pH samples were natural soils beneath the tailings and zone of amendment incorporation and the decline in concentrations could not necessarily be attributed to amendment incorporation. These results suggest that higher lime rates (as CaO and/or $\text{Ca}(\text{OH})_2$) would be necessary to reduce the pore water concentrations of these metals and reduce the risk of movement to groundwater. However, the near-neutral pH values and lower pore water metal levels in samples collected from natural soils beneath tailings indicates that the soil has been effective at buffering pH and attenuating metals in soil pore water leaching from the overlying tailings. This may explain some of the effectiveness of deep-plowing, which dilutes tailings with underlying natural soils during plowing.

Figure 4.2.34 shows the relative degree of oxidation in a soil system as a function of $pe + pH$ (Bohn et al 1985). Additionally, Figure 4.2.35 shows pe versus pH for samples for which metals were also analyzed. As pH increases, pe declines at a very steady rate. This is to be expected, as the concentration of protons decreases (pH increases) and suggests that both pH and Eh (pe) field data are dependable. The sum of ($pe + pH$) as shown in Figure 4.2.35 ranges between 16 and 18, indicative of relatively strongly oxidizing conditions (Bohn et al 1985) throughout the tailings/soil profile. This environment would be conducive to immobilization of metals at near-neutral pH levels through the formation of carbonate, sulfate, hydroxide and oxide solid phases (Lindsay 1979).

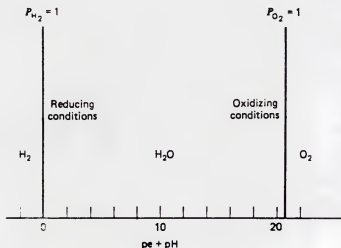


Figure 4.2.34. The range of oxidizing and reducing conditions in soil as defined by the breakdown of water to H_2 or O_2 (after Bohn et al 1985).

STARS: SILVER BOW CREEK RI/FS **pH and pe in Pore Water (1992)**

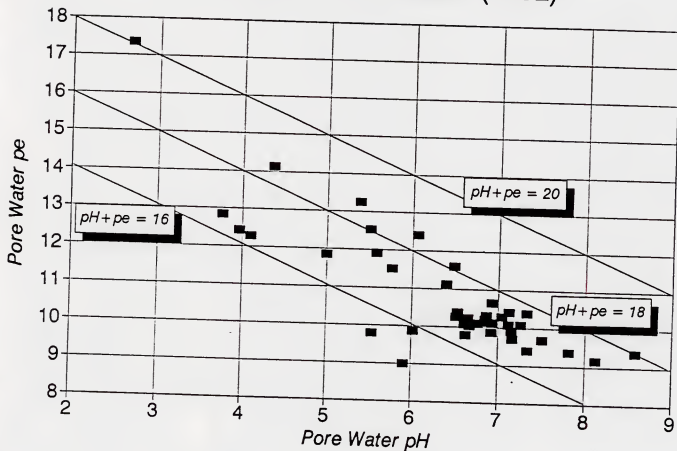


Figure 4.2.35. Plot of pe vs pH for 1992 pore water samples.

Major Cations

Figure 4.2.36 indicates that Ca and Mg are the dominant cations at all pH values in pore water, with concentrations ranging from 10^{-3} to approximately $10^{-4.5}$. Concentration is not affected by pH. Sodium is approximately 1 to 2 orders of magnitude lower than Ca, and increases from $10^{-6.5}$ to $10^{-4.5}$ as pH increases from 4 to 6. Potassium shows similar behavior as Na, but concentrations are generally about 2 orders of magnitude lower. The lower concentration of the monovalent cations, which do not bond as strongly to soil material as divalent cations, is typical of natural soils and is likely due to a higher degree of leaching relative to divalent cations (Hillel 1980).

Major Anions

Sulfate and bicarbonate are the dominant anions in soil pore water, with SO_4 comprising the majority of anions at low pH and bicarbonate increasing to levels similar to sulfate (10^{-3}) at neutral pH values (Figure 4.2.37). This is to be expected since the concentration of bicarbonate is probably controlled by soil CO_2 -carbonate equilibrium. Concentrations of other anions appear to be relatively unaffected by pH, typically deviating less than an order of magnitude with increasing pH. This is especially true of F above pH 5, which is similar in behavior to NO_3 , although NO_3 concentrations are approximately 2 orders of magnitude greater. Chloride concentrations range between 10^{-2} and 10^{-3} while PO_4 exhibits the lowest concentrations of any anion, increasing slightly at pH 6.5.

Aluminum

Aluminum concentrations show a decline of approximately an order of magnitude from pH 3 to 5.5, at which point concentrations drop sharply to approximately 10^{-7} (Figure 4.2.38), most likely in response to the formation of Al hydroxides and oxides or hydroxy-sulfates (Lindsay 1979). It should be pointed out that several points between pH 6 and 8 were below detection limit. It is also worth noting that the high Al concentration for the amended sample at pH 4 was collected from the 40 cm depth on the injection plot at site 7 and is likely the result of poor mixing associated with this incorporation technique, as discussed in earlier sections. This trend is also apparent in the solubility plots of the other metals.

Iron

Iron concentrations decreased approximately an order of magnitude from pH 4 to pH 5.5 (Figure 4.2.39), exhibiting a response similar to Al associated with the formation of oxides and hydroxides. In general, amendment incorporation decreased Fe levels to those found in the underlying natural soils. The single elevated concentration of Fe at pH 4 was collected from the 90 cm lysimeter on the ag-till plot at site 7. Recall that this lysimeter was located in a "hot spot" which yielded anomalously high metal and low pH samples throughout the study. Elevated metal levels associated with this lysimeter are apparent on the other plots.

Cation Activity in Pore Water (1992)

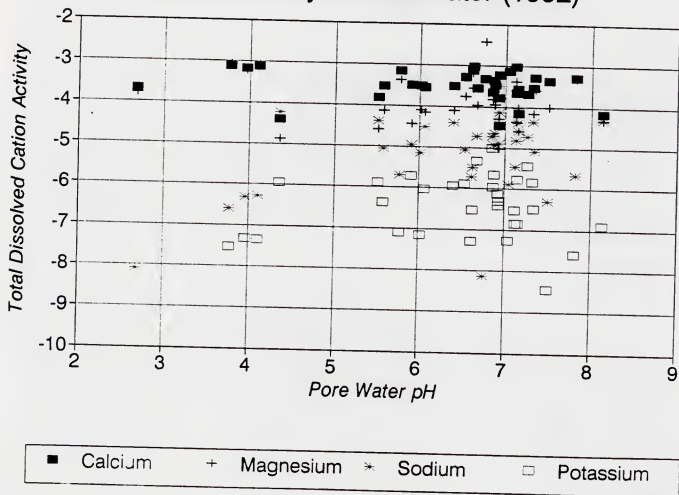


Figure 4.2.36. Solubility diagram for 1992 pore water major cations vs pH.

Anion Activity in Pore Water (1992)

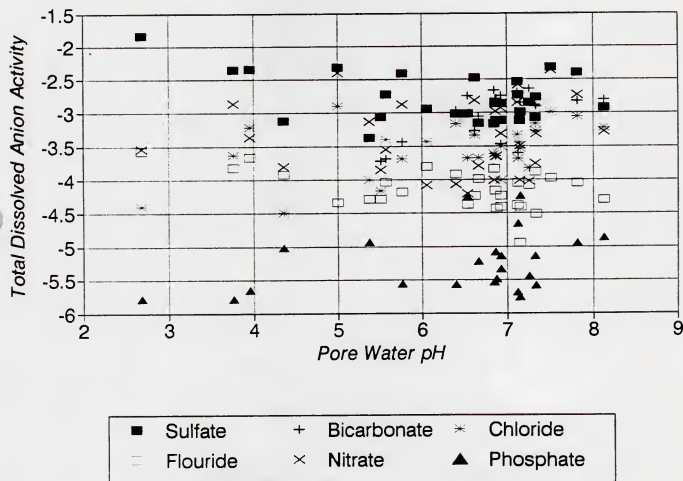


Figure 4.2.37. Solubility diagram for 1992 pore water major anions vs pH.

Aluminum Activity in Pore Water (1992)

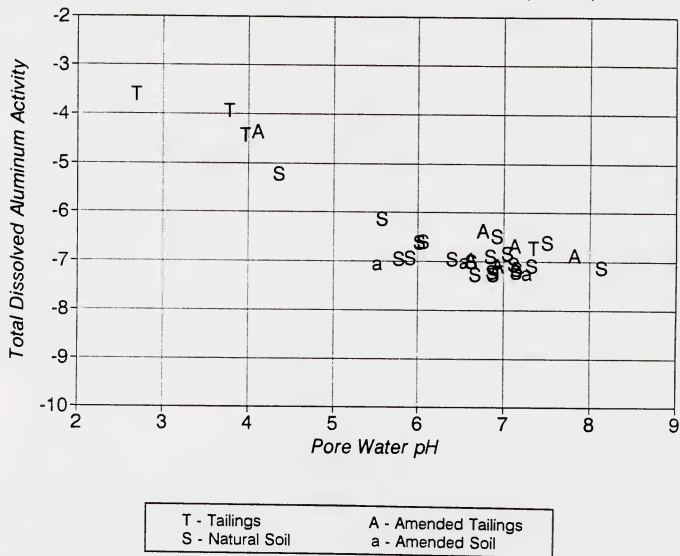
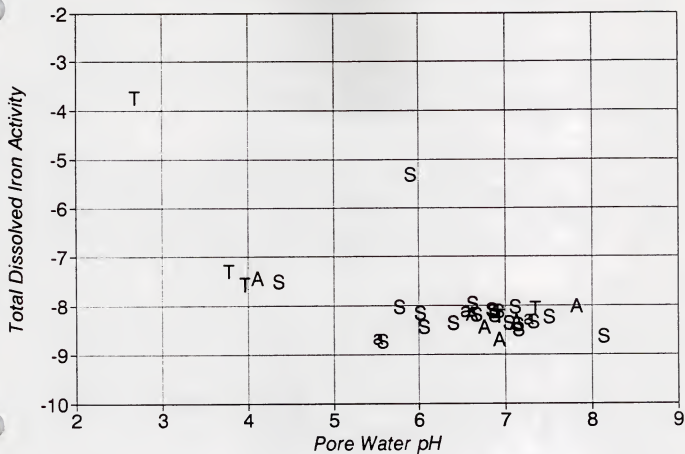


Figure 4.2.38. Solubility diagram for 1992 pore water aluminum vs pH.

Iron Activity in Pore Water (1992)



T - Tailings	A - Amended Tailings
S - Natural Soil	a - Amended Soil

Figure 4.2.39. Solubility diagram for 1992 pore water iron vs pH.

Manganese

Manganese exhibits a wide range of concentrations from pH 5 to 8, indicating that the concentration varies widely between sites and materials (Figure 4.2.40). For example, natural soils exhibit a wide range of concentrations both between sites and at an individual site. Also, some of the lower Mn values are associated with control plots at site 33. Although Mn concentrations on treated plots at this site may also be low, the low concentration is not necessarily attributed to the treatment effect. This same trend is apparent for Zn, and to a lesser degree Cd.

Although treatment increased the pH of tailings from 3-4 to 6-7, Mn concentrations were relatively unaffected. At pH 7 to 8, however, Mn concentrations decreased approximately 3 orders of magnitude. These data agree well with thermodynamic data and a large body of literature relating to the treatment of acidic, metal-rich mine waste, which indicate that very little decrease in Mn concentrations are obtained until a pH of approximately 8 to 9 is obtained, at which point Mn hydroxides and oxides form (Lindsay 1979).

Copper

Copper concentrations decreased roughly an order of magnitude between pH 3 and 5.5 (Figure 4.2.41). At pH 5.5 concentrations dropped considerably in all samples, declining to 10^{-7} to 10^{-8} at pH 7, an overall drop of approximately 4 orders of magnitude. The rapid decrease in concentrations at pH 5.5 is probably in response to the precipitation of Cu carbonate or hydroxide minerals. The low ($< 10^{-5}$) concentrations of Cu in pore water for all types of samples at pH values > 6 indicate that treatment of tailings with lime will provide an effective control on the movement of Cu to groundwater.

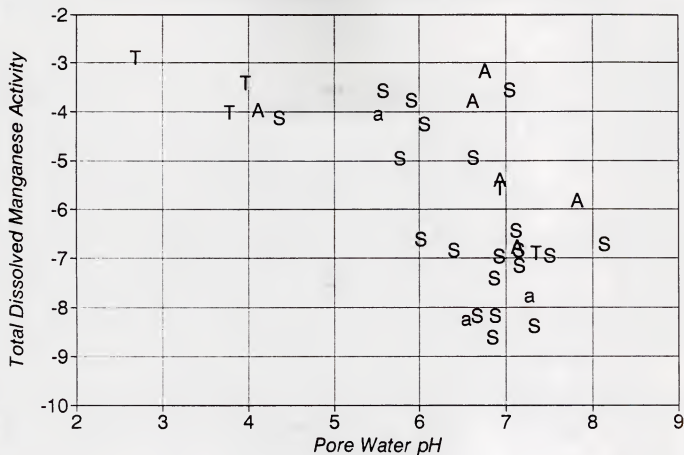
Cadmium

Figure 4.2.42 shows that the response of Cd to amendment incorporation was similar to that of Mn, where pH was increased to neutral values but not always with an attendant decrease in Cd concentrations, indicating that Cd mobility may be problematic at treated sites using STARS technology. As with Mn, there was also considerable variation in Cd concentrations between sites and materials. Concentrations of Cd generally showed a decrease of about 2 orders of magnitude above a pH of 7.

Lead

Initial Pb concentrations were relatively low (10^{-8} to 10^{-9}), even at low pH values, in tailings prior to amendment incorporation (Figure 4.2.43). However, the incorporation of amendment decreased Pb levels further by approximately one order of magnitude, to levels similar to those found in the underlying natural soils. This decrease occurred at a pH of about 6. Above this value, concentrations of Pb in pore water were relatively consistent. Many of the points in this range were below detection levels. These results indicate that Pb poses little threat to groundwater through potential leaching.

Manganese Activity in Pore Water (1992)

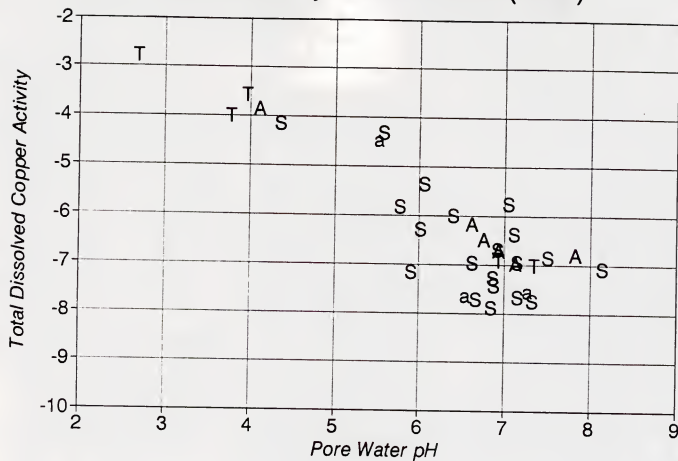


T - Tailings
S - Natural Soil

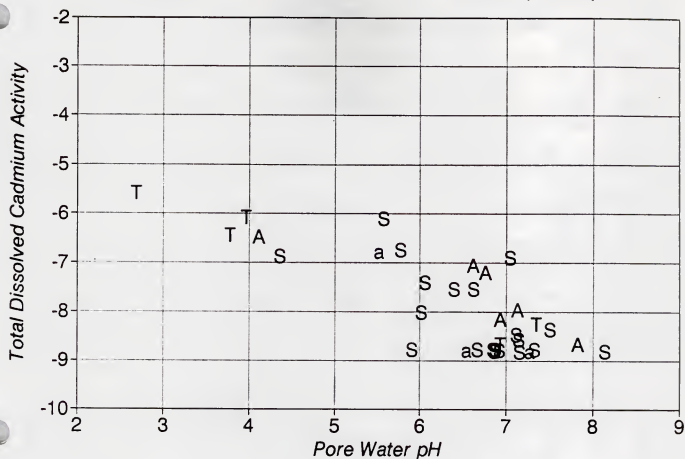
A - Amended Tailings
a - Amended Soil

Figure 4.2.40. Solubility diagram for 1992 pore water manganese vs pH.

Copper Activity in Pore Water (1992)



Cadmium Activity in Pore Water (1992)



T - Tailings

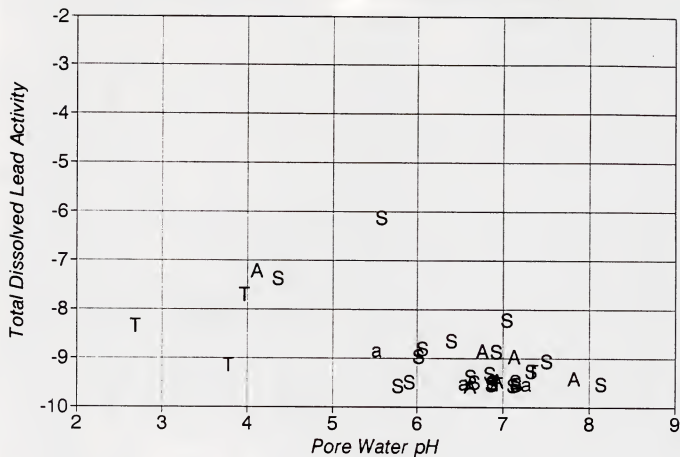
S - Natural Soil

A - Amended Tailings

a - Amended Soil

Figure 4.2.42. Solubility diagram for 1992 pore water cadmium vs pH.

Lead Activity in Pore Water (1992)



T - Tailings
S - Natural Soil
A - Amended Tailings
a - Amended Soil

Figure 4.2.43. Solubility diagram for 1992 pore water lead vs pH.

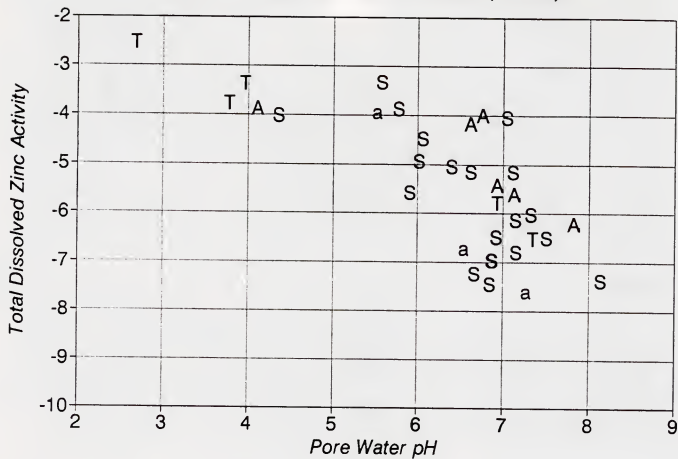
Zinc

The geochemical behavior of Zn in pore water was very similar to Mn both in treatment response by site and by pH. Generally, concentrations were lower in natural soils at site 33 (Volume III, Appendix B), as was the case for Mn. Although pH levels were increased to neutral values through amendment incorporation, Zn levels did not necessarily decrease (Figure 4.2.44). This may in part be due to the buffering effect of $\text{CaSO}_4 \cdot \text{H}_2\text{O}$. As lime solubilizes, the Ca is consumed by the formation of gypsum through the pairing of Ca with SO_4 , which is in ample supply in most cases. Because gypsum is more soluble than lime (CaCO_3), it will support greater concentrations of Ca in solution. The greater concentration of Ca results in a decrease in the concentration of CO_3 (as HCO_3) to a level below that which would be supported by CaCO_3 . This phenomenon is referred to as the "common ion effect" (Bohn et al 1985) and in this case, results in the maintenance of a lower pH (in the range of 6 to 7.5) until SO_4 concentrations decrease and come into equilibrium with gypsum. The result is that a considerable amount of lime would need to be added to increase the pH to a point where Zn concentrations would be reduced through the formation of Zn solid phases (carbonates, hydroxides and oxides). This is also true for Cd and Mn.

Arsenic

Pore water results indicate that unlike the metals, As solubility *increases* with increasing pH. These results fit well the thermodynamic data available for As which indicates an increase in solubility around pH 7 for both solid and sorbed phases of As (Lindsay 1979). At pH 4, As concentrations are at 10^{-7} , but increase to approximately 10^{-5} between pH 6 and 8 (Figure 4.2.45). The higher pH of the natural soils and amended tailings results in greater pore water As concentrations and indicates that As may be difficult to control using STARS technology.

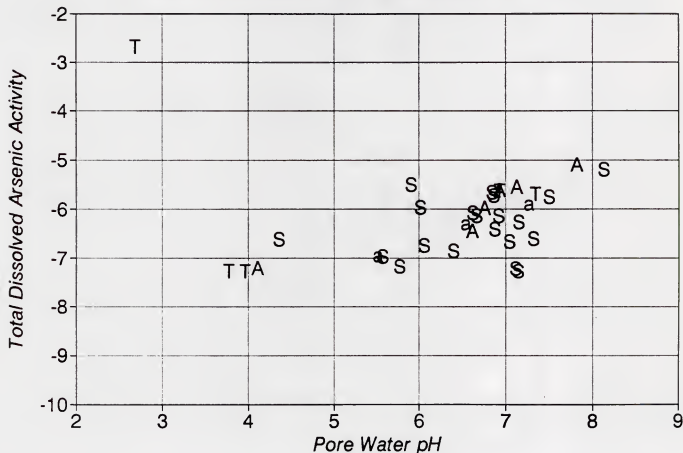
Zinc Activity in Pore Water (1992)



T - Tailings	A - Amended Tailings
S - Natural Soil	a - Amended Soil

Figure 4.2.44. Solubility diagram for 1992 pore water zinc vs pH.

Arsenic Activity in Pore Water (1992)



T - Tailings
S - Natural Soil
A - Amended Tailings
a - Amended Soil

Figure 4.2.45. Solubility diagram for 1992 pore water arsenic vs pH.

4.2.3 Extractable Metals

The ability of plants to accumulate metals is limited. In addition to the various soil characteristics that influence bioavailability, several plant factors also affect their ability to accumulate metals. These include plant species, cultivars, different plant parts (roots, vegetative tissues, seeds), plant age, micronutrient interactions, fertilization practices, and climatic factors. Attempts to quantify the bioavailability and mobility (to vegetation) of metals have led to the very extensive use of numerous soil extracts. The extractable concentrations may provide some indication of the bioavailability of the metals. Extracts in common use include ammonium acetate, dilute acids, neutral salts, and various organic chelators. In this treatability study, the extractable (bioavailable) metal fractions in the amended wastes/soils were measured in ammonium acetate extract solutions of the collected samples. The extract used for As was the Bray P-1 solution.

In 1991, samples of amended wastes/soils were collected from each experimental plot within two replications at STARS Sites 7, 21, 27 and 33. Since the seeded vegetation did not survive in 1991 at Site 2 (see Section 4.3), no samples were obtained. The amended wastes/soils were collected from two depth increments, 0-15 cm and 15-46 cm, using a hand auger. These two depths were chosen because the majority of vegetation roots are within the top 46 cm of the profile. The samples were air-dried, sieved to pass a 2 mm screen, and extracted using ammonium acetate (pH 7.0, 1N). A separate extraction using the Bray P-1 solution was also performed. The extract solutions were analyzed for concentrations of As (Bray extract) and Cd, Cu, Pb, Mn and Zn (ammonium acetate extract).

The mean extractable metal and As concentrations in amended wastes/soils are shown in Tables 4.2.7 through 4.2.10. The data are arranged by STARS treatment, analyte, and depth of amended wastes/soils. The mean data were not statistically compared because only two replications were represented in the sampling and analysis scheme.

Extractable concentrations found in these materials at Site 7, Rocker (Table 4.2.7) were similar both across treatments and within the profile. This was markedly different than the orders of magnitude change in concentrations found in the water soluble data for several metals.

Table 4.2.7. Mean extractable¹ metal concentrations (mg/kg) in amended wastes/soils at STARS Site 7.

Amended Waste/ Soil Depth (cm)	Treatment				
	Control	Ag Till	Deep Plow	Injection	Coversoil
--- Arsenic ---					
0 - 15	261 ²	202	205	91	158
15 - 46	186	214	337	208	188
--- Cadmium ---					
0 - 15	0.41	0.29	0.54	0.12	0.63
15 - 46	0.50	0.53	0.45	0.25	0.71
--- Copper ---					
0 - 15	78	41	73	14	130
15 - 46	112	91	88	91	290
--- Lead ---					
0 - 15	0.42	0.93	3.18	0.35	3.08
15 - 46	2.15	3.65	8.76	0.58	175
--- Zinc ---					
0 - 15	25	17	24	3.0	44
15 - 46	45	55	30	15	69

¹ Bray P-1 extract for As; ammonium acetate (pH 7, 1N) for Cd, Cu, Pb, and Zn.

² Mean values based on N = 2.

Table 4.2.8 exhibits extractable metals and As data for materials at Site 21 (Ramsay Flats). These data are variable and no apparent trends could be discerned across the STARS treatments or by profile depth. Extractable or bioavailable concentrations of As, Cd, Cu, Pb, and Zn in amended wastes/soils at the Fairmont Site (27) were generally elevated in the 0-15 cm depth compared to the 15-46 cm layer. The materials in the treated plots also seemed to have lower levels compared to the wastes/soils in the control plot (Table 4.2.9). These two general trends were demonstrated for extractable levels in the samples collected from Site 33, Opportunity (Table 4.2.10).

Table 4.2.8. Mean extractable¹ metal concentrations (mg/kg) in amended wastes/soils at STARS Site 21.

Amended Waste/ Soil Depth (cm)	Treatment				
	Control	Ag Till	Deep Plow	Injection	Coversoll
--- Arsenic ---					
0 - 15	1031 ²	57	16	32	106
15 - 46	215	146	38	632	153
--- Cadmium ---					
0 - 15	3.25	2.15	2.55	1.26	1.80
15 - 46	4.60	2.85	4.45	0.80	4.05
--- Copper ---					
0 - 15	423	193	765	231	238
15 - 46	1240	798	1290	87	1205
--- Lead ---					
0 - 15	0.06	0.03	64	0.07	0.13
15 - 46	1044	167	741	1.40	785
--- Zinc ---					
0 - 15	559	158	149	108	116
15 - 46	725	214	278	37	267

¹ Bray P-1 extract for As; ammonium acetate (pH 7, 1N) for Cd, Cu, Pb, and Zn.

² Mean values based on N = 2.

The purpose for obtaining these bioavailable metal data was to ascertain if they could be related to metal levels in the vegetation grown on the experimental plots. Section 4.3 of this report provides a detailed discussion of the STARS vegetation monitoring including exhibits of metal concentrations for the dominant plant species that grew at each site. Correlations between soil extractable (bioavailable) metal and As levels and those found for vegetation were calculated. It should be noted that metal and arsenic levels reported for the field vegetation represent total plant loadings. That is, the metals and arsenic are both on the plants and incorporated in them. The pairs used in the statistical analysis were the depth integrated mean bioavailable concentrations for a specific experimental plot and the concentrations for the species-specific plants growing in that plot. Correlation coefficients (r value) are exhibited for each analyte in Table 4.2.11. The data are arranged by STARS

Table 4.2.9. Mean extractable¹ metal concentrations (mg/kg) in amended wastes/soils at STARS Site 27.

Amended Waste/ Soil Depth (cm)	Treatment			
	Control	Ag Till	Deep Plow	Extra
--- Arsenic ---				
0 - 15	306 ²	265	171	430
15 - 46	111	119	109	170
--- Cadmium ---				
0 - 15	6.35	2.40	1.65	1.35
15 - 46	2.65	1.10	0.69	1.05
--- Copper ---				
0 - 15	449	50	18	101
15 - 46	77	7.0	3.5	8.7
--- Lead ---				
0 - 15	969	19	10	18
15 - 46	75	8.4	3.7	7.2
--- Zinc ---				
0 - 15	612	41	46	24
15 - 46	271	66	32	61

¹ Bray P-1 extract for As; ammonium acetate (pH 7, 1N) for Cd, Cu, Pb, and Zn.

² Mean values based on N = 2.

site, dominant species grown at the site, number of bioavailable/vegetation concentration pairs and the calculated correlation coefficients. Of the 60 correlations, 17 were found to be statistically significant ($p \leq 0.05$), with the wheatgrass species having more significant correlations compared to the wildrye grass species. No significant correlations were exhibited for As.

To further evaluate the relationships between bioavailable concentrations in the amended wastes/soils and levels found for the vegetation, these data were combined by wheatgrass species, by wildrye grass species, and then using all plant values. Correlations are graphically displayed in Figures 4.2.10 through 4.2.12. Arsenic found in ryegrass plants was positively correlated to the measured bioavailable waste/soil As levels. The correlation

Table 4.2.10. Mean extractable¹ metal concentrations (mg/kg) in amended wastes/soils at STARS Site 33.

Amended Waste/ Soil Depth (cm)	Treatment				
	Control	Ag Till	Deep Plow	Injection	Coversoil
--- Arsenic ---					
0 - 15	88 ²	28	26	23	25
15 - 46	63	52	35	39	33
--- Cadmium ---					
0 - 15	1.65	0.92	0.38	0.22	0.36
15 - 46	1.85	1.30	0.48	0.27	0.74
--- Copper ---					
0 - 15	316	139	46	41	34
15 - 46	307	175	70	54	75
--- Lead ---					
0 - 15	0.48	0.38	0.29	0.05	0.17
15 - 46	7.10	3.7	0.59	0.49	0.56
--- Zinc ---					
0 - 15	227	29	9.6	9.4	12
15 - 46	208	62	12	9.6	36

¹ Bray P-1 extract for As; ammonium acetate (pH 7, 1N) for Cd, Cu, Pb, and Zn.

² Mean values based on N = 2.

($r = 0.638$) was significant at the $p \leq 0.05$ level. The amount of As found in wheatgrass samples was negatively correlated (nonsignificant) to the extractable waste/soil concentrations (Figure 4.2.46, top). The reason(s) for the difference in correlation between the wheatgrass and the wildrye grasses is unknown. When the data for As were combined, the r value was -0.082 indicating no relationship between bioavailable As as measured in Bray P-1 extract of the amended wastes/soils and As found in the vegetation grown in the test plots.

Relationships for the metals measured in the ammonium acetate extract and metals in the vegetation are shown in Figure 4.2.46, bottom (Cu), Figure 4.2.47 (Cd and Pb), and

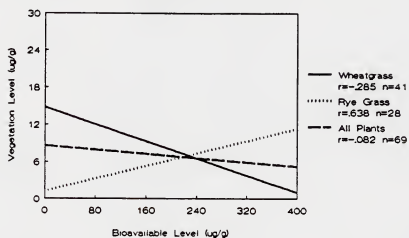
Table 4.2.11. Correlation between soil extractable bioavailable metal levels and vegetation metal levels for each species at each site.

Site	Dominant Plant Species	N ¹	Statistic ²	As	Cd	Analyte			
						Cu	Pb	Mn	Zn
07 (Rocker)	Thickspike wheatgrass (<i>Agropyron dasystachyum</i>)	8	r	.464	.869	.651	.232	-.568	.960
	Altai wildrye (<i>Elymus augustus</i>)	8	r	.521	.676	.731	.071	-.280	.634
21 (Ramsay Flats)	Tall wheatgrass (<i>Agropyron elongatum</i>)	8	r	-.433	.742	.803	.958	.673	.925
	Intermediate wheatgrass (<i>Agropyron intermedium</i>)	8	r	-.412	.526	.481	.727	.554	.764
27 (Fairmont)	Crested wheatgrass (<i>Agropyron cristatum</i>)	6	r	.423	-.438	.852	.211	.852	.341
	Desert wheatgrass (<i>Agropyron desertorum</i>)	6	r	.790	.376	.937	.531	.938	.592
	Russian wildrye (<i>Elymus junceus</i>)	6	r	.563	.268	.738	.879	.670	.536
33 (Opportunity)	Streambank wheatgrass (<i>Agropyron riparium</i>)	5	r	.737	.968	.347	.236	.902	.372
	Altai wildrye (<i>Elymus augustus</i>)	7	r	.664	.817	.654	.594	.842	.910
	Basin wildrye (<i>Elymus cinereus</i>)	8	r	.472	.317	.139	.106	-.210	.274

¹ N is number of soil/vegetation pairs.

² r is correlation coefficient; shaded values are significant at $P \leq 0.05$.

Correlations between soil and plant Arsenic



Correlations between soil and plant Copper

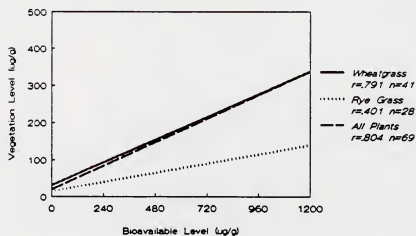
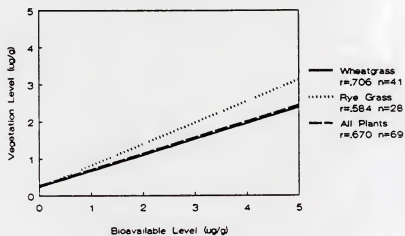


Figure 4.2.46. Relationship between soil/plant As (top) and Cu (bottom) levels.

Correlations between soil and plant
Cadmium



Correlations between soil and plant
Lead

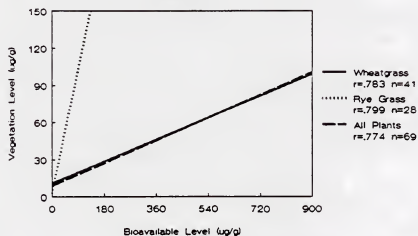
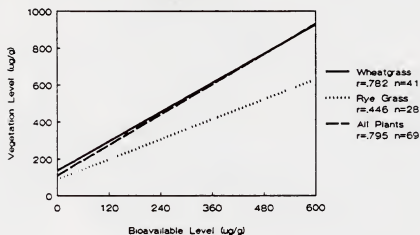


Figure 4.2.47. Relationship between soil/plant Cd (top) and Pb (bottom) levels.

Correlations between soil and plant Manganese



Correlations between soil and plant Zinc

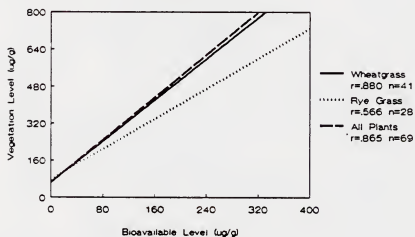


Figure 4.2.48. Relationship between soil/plant Mn (top) and Zn (bottom) levels.

Figure 4.2.48 (Mn and Zn). All correlations were positive and statistically significant ($p \leq 0.05$).

Data for extractable metals supporting tables and figures in this section are presented in Volume III, Appendix B2.

4.2.4 Total Metals

In May of 1992, amended wastes/soils representing the surface (0-2.5 cm) layer were collected from three locations in every experimental plot at STARS Sites 7, 21, 27, and 33. The three surface material samples were composited into one sample. Samples were also obtained from locations outside, but adjacent, to the experimental plots. These represented the *no action* alternative and are different than the control plots which were tilled, fertilized, seeded, and mulched. The samples were air-dried, and materials passing a 2 mm screen were sent to a laboratory for quantification of levels of As, Cd, Cu, Pb, and Zn using x-ray fluorescence (XRF) spectroscopy.

The purpose for obtaining these data was to allow the Operable Unit Feasibility Study team to evaluate and compare the STARS technology to other potential alternatives in terms of the direct contact and potential fugitive dust pathways.

Mean total metal and As concentrations in amended wastes/soils are shown in Tables 4.2.12 through 4.2.15. Data are arranged by analyte and STARS treatment. The coversoil mean values are in the materials above the wastes and represent concentrations in the imported soil. Total concentrations for material at Site 7 (Rocker) indicated few significant differences among the treatments for any analyte. Mean levels in the imported coversoil are lower than the amended or unamended wastes (Table 4.2.12). Concentrations of these metals in the control, no action, and the amended wastes are statistically identical for As, Cu, Pb, and Zn.

Total metal and As levels for materials at Ramsay Flats (21) are exhibited in Table 4.2.13. Absolute concentrations are much higher than those found at the Rocker Site. The treated materials had the same mean As concentrations as the control and no action alternative wastes, while Cd concentration was significantly elevated in the undisturbed wastes collected from the no action alternative areas. Concentrations of the other metals revealed significant treatment differences, but no patterns were discerned.

Mean total metal concentrations in amended wastes/soils at the Fairmont Site (27) are shown in Table 4.2.14. The deep plow treatment significantly reduced total As, Cd, Cu, and Pb concentrations compared to the agricultural tillage and extra treatments and the unamended control and no action alternative. The method of contamination at this site was overland flow of irrigation water from Silver Bow Creek. It is perceived that contamination

Table 4.2.12. Mean total metal concentrations (mg/kg) in amended wastes/soils determined by XRF at STARS Site 7. Data for surface soils (0-2 cm).

Element	Treatment ¹					
	Control	Ag Till	Deep Plow	Injection	Coversoil	No Action
Arsenic	285 B ²	791 B	288 B	283 B	14 A	290 B
Cadmium	4.0 A	6.7 B	4.2 A	4.8 AB	4.0 A	5.6 AB
Copper	468 B	791 B	764 B	484 B	50 A	468 B
Lead	325 B	447 B	533 B	333 B	47 A	285 B
Zinc	881 B	1016 B	1489 B	1000 B	132 A	1035 B

¹ N = 4.

² Multiple mean comparisons based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

Table 4.2.13. Mean total metal concentrations (mg/kg) in amended wastes/soils determined by XRF at STARS Site 21. Data for surface soils (0-2 cm).

Element	Treatment ¹					
	Control	Ag Till	Deep Plow	Injection	Coversoil	No Action
Arsenic	1671 B ²	1927 B	1524 B	1282 B	101 A	1216 B
Cadmium	24.5 C	25.7 CD	21.0 CB	17.4 B	4.0 A	34.3 D
Copper	4519 CB	4592 CB	4285 CB	2691 B	419 A	4954 C
Lead	2075 B	1986 B	3491 C	1479 B	167 A	1730 B
Zinc	7194 CB	7980 CB	6637 CB	5432 B	845 A	8472 C

¹ N = 4.

² Multiple mean comparisons based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

at this type of site is limited to the surface layer. Tillage by agricultural equipment should reduce surface concentrations because of dilution. This is supported by lower (not always significant) levels of these contaminants in the agricultural tillage and extra plots compared to the control and no action waste concentrations. It is apparent that deep tillage reduced total metal and As levels by nearly an order of magnitude compared to the no action alternative levels.

The coversoil material at Site 33 had significantly lower concentrations of As, Cu, Pb, and Zn compared to the other amended or unamended wastes (Table 4.2.15). There were no differences among the amended wastes (ag-till, deep plow, and injection) compared to the untreated wastes (control and no action alternative).

Surface chemistries observed in this one sampling in the spring quantify the total metals and As at a specific time and within a specific hydrologic regimen. The formation of soluble salts on the surface of streamside tailings is well known and is considered to be a significant pathway of contaminant movement into surface waters. Simulated rainfall/runoff tests (Section 4.1.4) were conducted to provide information about movement via this pathway. As a consequence, total surface material metal levels may fluctuate as a function of chemical, physical, and environmental factors.

Data for surface soil total metals supporting tables and figures in this section are presented in Volume III, Appendix B5.

Table 4.2.14. Mean total metal concentrations (mg/kg) in amended wastes/soils determined by XRF at STARS Site 27. Data for surface soils (0-2 cm).

Element	Treatment ¹				
	Control	Ag Till	Deep Plow	Extra	No Action
Arsenic	598 B ²	331 B	108 A	400 B	646 B
Cadmium	17.6 CB	11.2 B	5.8 A	10.6 B	23.7 C
Copper	2931 CB	1303 B	447 A	1596 CB	3819 C
Lead	3802 CB	1349 B	419 A	1445 B	5821 C
Zinc	5861 B	1914 A	733 A	1986 A	8551 B

¹ N = 4.

² Multiple mean comparisons based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

Table 4.2.15. Mean total metal concentrations (mg/kg) in amended wastes/soils determined by XRF at STARS Site 33. Data for surface soils (0-2 cm).

Element	Treatment ¹					
	Control	Ag Till	Deep Plow	Injection	Coversoil	No Action
Arsenic	252 B ²	309 B	298 B	257 B	32 A	200 B
Cadmium	6.9 AB	8.5 B	6.3 AB	4.1 A	5.2 AB	6.1 AB
Copper	1690 B	1556 B	1318 B	966 B	133 A	1309 B
Lead	524 B	617 B	646 B	505 B	79 A	412 B
Zinc	2188 B	2399 B	2009 B	1560 B	300 A	1698 B

¹ N = 4.

² Multiple mean comparisons based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

4.2.5 Amendment Mixing

Excavations were made to a depth of approximately five feet in a total of 10 deep plow and lime slurry injection plots to determine which incorporation technique most evenly distributed the amendment material throughout the treated interval, to define the depth of incorporation and to compute the volume of amended material. For comparison, excavations were also made in control plots at sites 02 and 21, and in the agricultural incorporation treatment plot at site 21. A pH indicator dye was sprayed on the face of the excavation followed by the application of a white powder to intensify the color (Section 3.1.2). Photographs were taken of the pit walls prior to and after the pH indicator dye was applied and are included in Appendix Volume II, C4. The photographs below in Figure 4.2.49 through Figure 4.2.54 were chosen to represent each of the amendment incorporation methods.

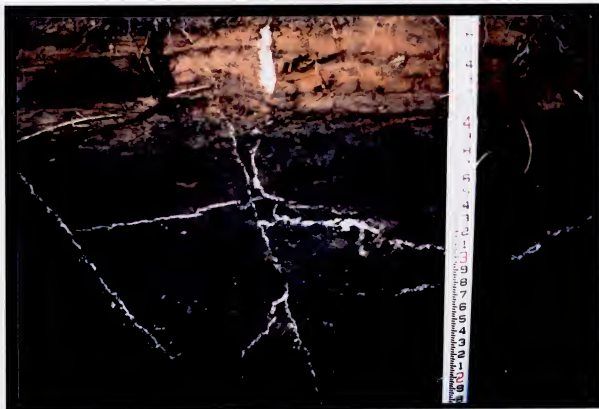


Figure 4.2.49 - Detailed photograph of the lime slurry injection treatment excavation taken prior to application of the pH indicator dye (21104).



The injected lime slurry tended to follow only the injection bores and move into the soil profile through the largest soil pore channels, creating vertical and lateral lime stringers (Figure 4.2.49). Additionally, due to back pressure when impermeable materials were encountered, some of the slurry was rejected from the injection bores and ponded on the surface. Rototilling of the rejected material into the upper eight inches of the soil profile generally elevated the pH of this layer above 8.5, while much of the deeper profile remained largely untreated (Figure 4.2.50).

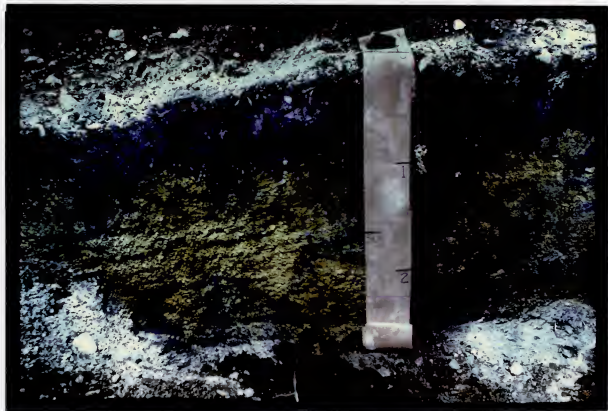


Figure 4.2.50 - Lime slurry injection treatment excavation with pH indicator (07108).

The proposed depth of the lime slurry injection treatment was 4.0 feet. Indeed, injection tubes were pushed into the soil profile the full four feet and therefore, some slurry may be found at that depth. However, distribution and uniform mixing of the amendment material appear to be poor. The depth of excavations at some sites, such as at site 07, were limited by high ground water conditions.



A detailed photo of a deep plow excavation is shown below in Figure 4.2.51. Excavations of deep plow plots clearly showed a more even distribution of amendments (compared to injection plots) and a strong preference of plant roots to follow zones of high pH soil (Figure 4.2.51).

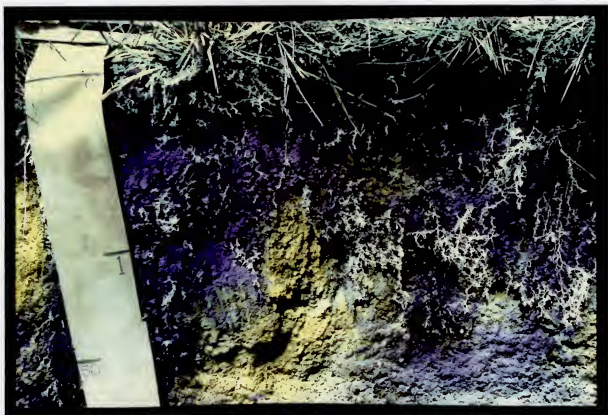


Figure 4.2.51 - Detailed photo of a deep plow excavation with pH indicator applied (07D09).

A deep plow plot excavation is shown below in Figure 4.2.52. Incorporation of amendment material deeper into the soil profile and more uniform mixing is evident by the purple zones (pH >6.5) throughout the soil profile. The deep plow incorporation treatment did not impact material below 3.5 feet. The purple zones in these areas are assumed to be undisturbed native material, below the zone of incorporation.

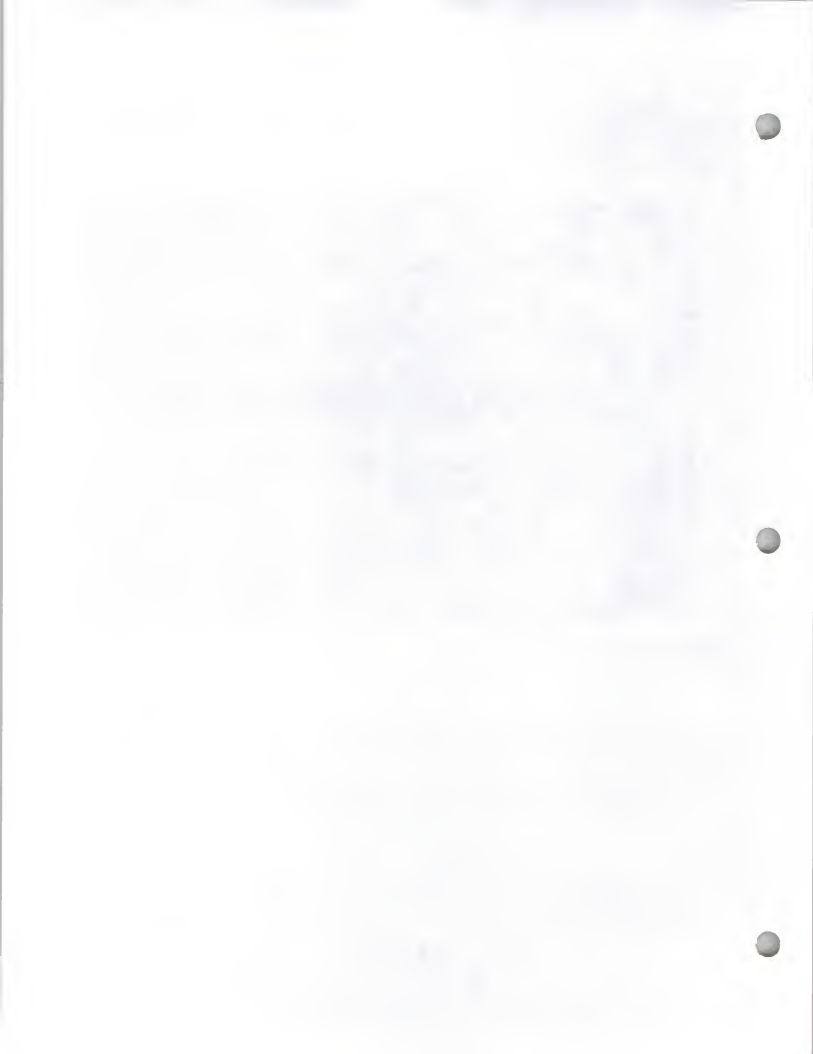
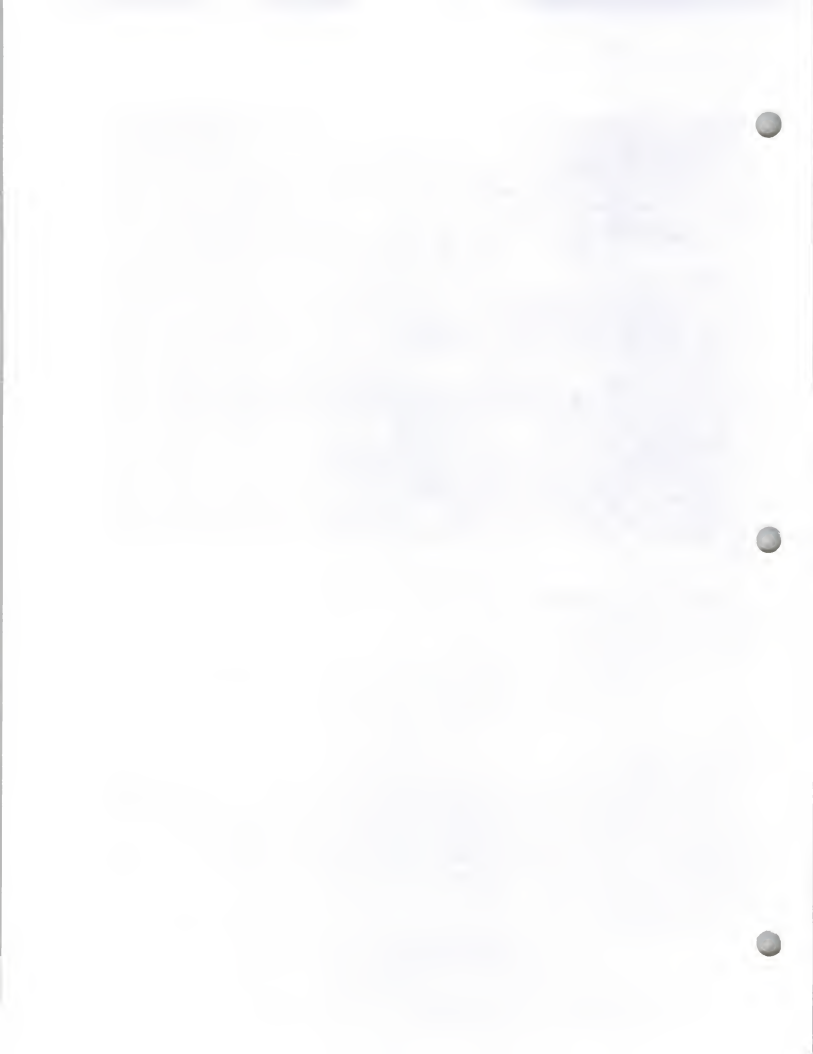




Figure 4.2.52 - Deep plow treatment excavation with pH indicator (21D02).

The lighter colored areas in Figure 4.2.52 appear to be sections of the soil profile where the pH indicator was inadvertently not applied or not applied heavily enough. Because of the surrounding amended material, it is believed that these areas have also been amended and should show a purplish color, indicating a pH of 6.5 or above.

Figure 4.2.53 shows an ag-till treatment excavation with the pH indicator and white powder applied. The proposed depth of amendment incorporation for this treatment was eight inches. Material below this zone was not impacted by this amendment incorporation treatment. Amendment material was successfully incorporated to the proposed depth as shown by the purple color, indicative of a high pH level, across the upper soil profile. This method was successful in incorporating the amendment to the proposed depth of eight inches as well as uniformly distributing the amendment material in this zone. The ag-till treatment may be very useful for treating shallow tailings.



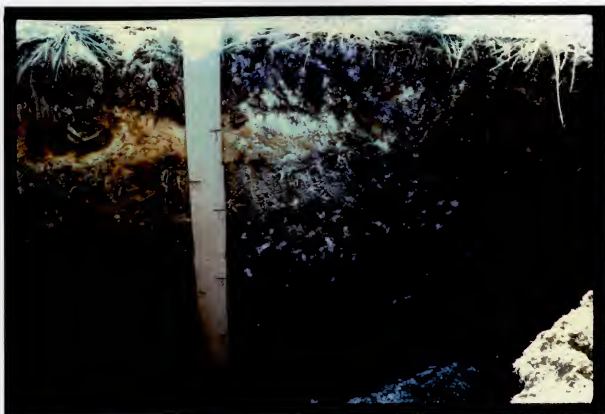


Figure 4.2.53 - Ag-Till treatment excavation with pH indicator (21A05).

For comparison, a control plot excavation is shown below in Figure 4.2.54. No pH indicator dye or white powder was applied in this cross section. Relic plant root growth is evident at the surface of the buried soil at approximately 1.5 foot depth.

The excavations also provided a means to describe the volume of tailings neutralized. The calculated percentages of the soil profile successfully treated ($\text{pH} > 6.5$) and exposed in each soil pit was determined by measuring areas of soil with a pH greater than 6.5 and areas with a $\text{pH} < 6.5$, to the depth of incorporation (Table 4.2.16). Amendment mixing volumes at each site are shown in Figure 4.2.55 through Figure 4.2.57. Due to poor quality photographs, no amendment mixing volume calculations could be made for Site 27 (Fairmont). In general, the percentage of the soil pit face area successfully treated ($\text{pH} > 6.5$) was best for the deep plow (62 to 76 percent). A greater range was evident for the injection treatment in which the success ranged from 29 to 66 percent.



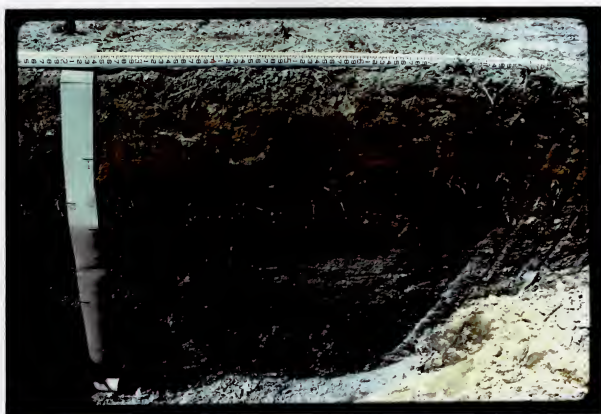


Figure 4.2.54 - Control plot excavation (21C06).

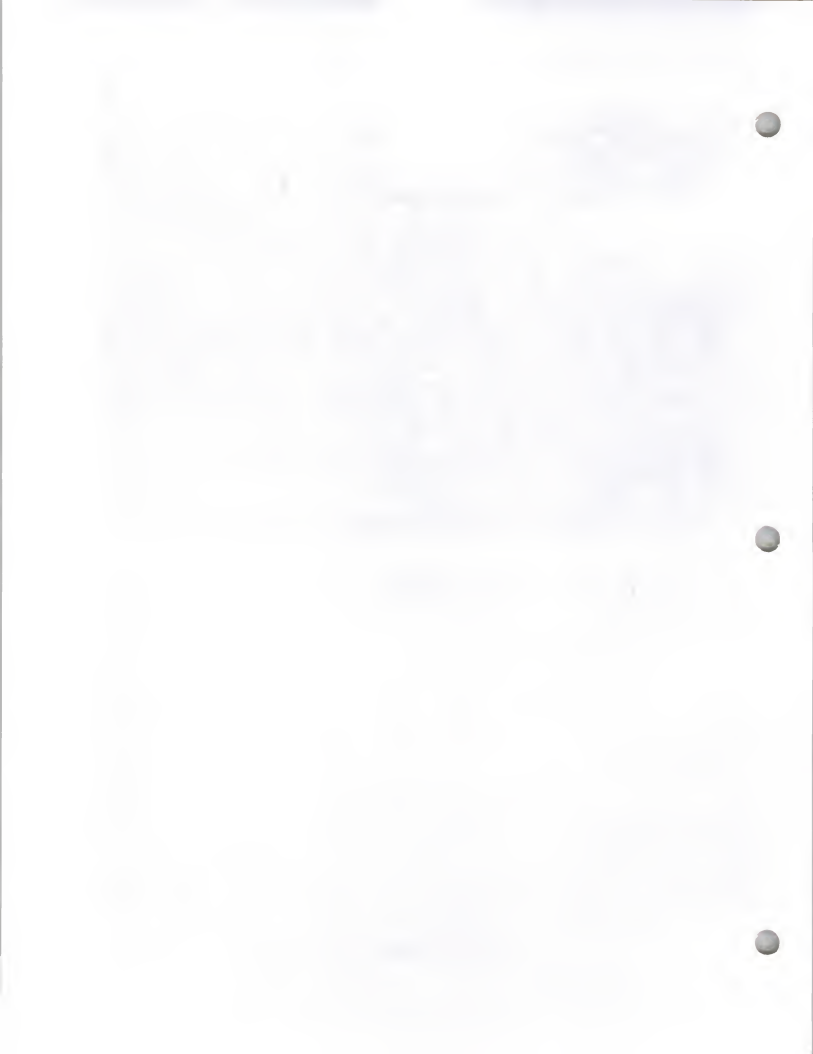


Table 4.2.16 - Amendment Mixing Computations from Photographs in Appendix Volume II, C4.

Plot	Proposed Incorporation Depth (ft)	Measureable Depth In Photo (ft)	Measureable Width In Photo (ft)	Total Area in photo (ft ²)	% R ¹¹	Area >6.5 pH	% of Total	Area <6.5 pH	% of Total	% U ⁹
02I03	4.0	3.5 ¹	7.2	25.2	36	7.3	29	17.9	71	-
02D06	3.5	2.0 ⁴	9.0	18.0	29	11.8	66	0.84	4	30
07I08	4.0	2.2 ³	4.75	10.45	15	5.05	48	4.2	40	11
07D09	3.5	2.4 ²	6.7	16.08	26	9.9	62	2.7	17	21
21A05	.67	.67	2.3 ⁵	1.54	13	1.1	71	.44	29	-
21D02	3.5	3.5	9.0	31.5	51	24.0	76	3.2	10	11
21I04	4.0	1.5 ⁶	4.0	6.0	.09	3.7	61	-	-	39
27D05 ⁸	3.5	-	-	-	-	-	-	-	-	-
33D23	3.5	3.5	5.0	17.5	29	12.6	72	-. ⁷	-	15/13 ¹⁰
33I13	4.0	4.0	8.0	32.0	46	21.0	66	2.34	7	27

¹ Depth measured from top of pit to groundwater

² Depth to groundwater

³ Depth to bottom of pit

⁴ Depth to bottom of pit

⁵ Width of pH indicator dye application

⁶ Depth of tailings = 1.5'; below 1.5' is native material (pH >6.5 prior to treatment)

⁷ Undetermined due to poor photo

⁸ Unuseable photographs-no data

⁹ Percentage of area in photo which is assumed to be unaffected by the treatment or which cannot be determined due to photo size or angle

¹⁰ 15% undetermined due to poor photo/13% of the area looks unaffected

¹¹ % of the total plot cross-section area represented by measurements in photo (100% would be cross-section area of photo encompassing both plot boundaries)

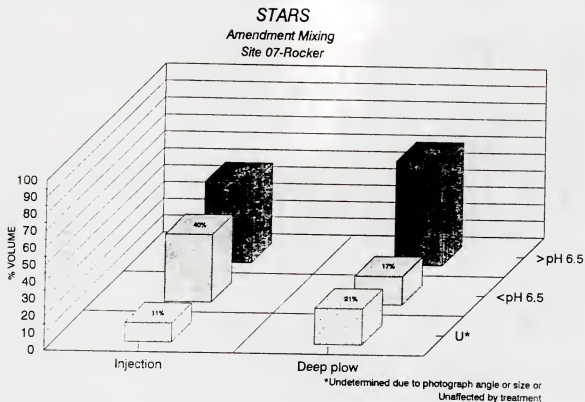


Figure 4.2.55 Amendment Mixing Volumes at site 07 (Rocker).

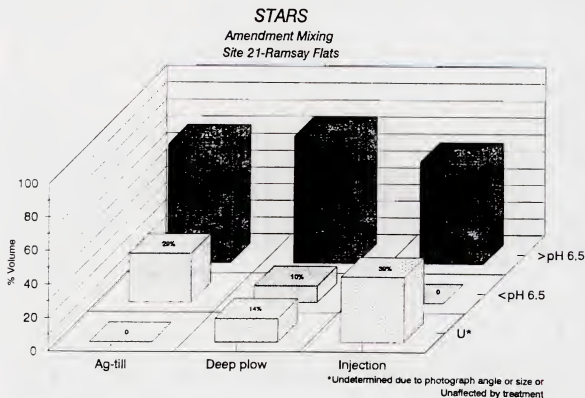
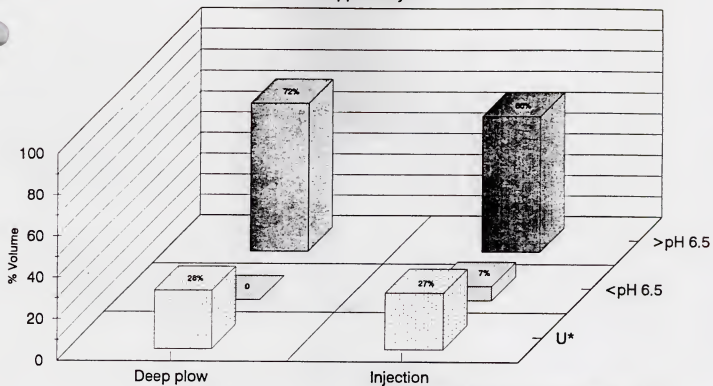


Figure 4.2.56 Amendment Mixing Volumes at site 21 (Ramsay Flats).

STARS
Amendment Mixing
Site 33-Opportunity



*Undetermined due to photograph angle or size or
Unaffected by treatment

Figure 4.2.57 Amendment Mixing Volumes at site 33 (Opportunity).

4.3 VEGETATION MONITORING

The goals of the vegetation portion of the treatability studies were to identify plant species that 1) are easy to establish; 2) have long-term survivability; 3) provide ample areal cover and production; and 4) do not accumulate (both in and on the plant) metals and arsenic in quantities sufficient to be harmful to grazing animals. The vegetation experimental design including species, seed mixes, seeding rates, and fertilization regimen are found in the STARS Phase III document (Reclamation Research Unit, Schafer & Associates and CH2M HILL, Inc. 1989). Table 4.3.1 shows the vegetation seeded or sprigged at each STARS site. Seeding and planting was carried out in mid-April of 1989 at Sites 7, 21, 27 and 33 and in mid-May 1989 at Site 2.

Specific STARS Phase III vegetation monitoring goals were to collect vegetation performance data and plant samples for metal and arsenic determinations. These data can provide the Feasibility Study team with sufficient information to design effective and appropriate revegetation programs and judge them against other potential remedial alternatives for the Streamside Tailings Operable Unit. Data measurements made during the course of monitoring included the following:

- Seeded species growth and development
 - seed germination (plant density) measured in 1989.
 - seedling survival evaluated during the first two growing seasons.
 - vegetation areal cover measured in 1989, 1990, and 1991.
 - vegetation production measured in 1991.
 - rooting characteristics including depth of penetration.
- Seeded species tissue analyses
 - determination of total Al, As, Cd, Cu, Pb, Mn, and Zn concentrations in dominant species.

Rapidly established and persistent stands of vegetation grown on the mine wastes along Silver Bow Creek should reduce the movement of contaminants into soil pore water, surface water, and into the atmosphere via fugitive dust (see conceptual model in Section 2.4). To determine whether the STARS remedial technology will promote a productive and persistent plant community, the current status (static) of vegetation was determined. Measurements included areal cover, peak standing crop, plant metal levels, and rooting depths.

Percent areal cover will provide evidence of the relative effectiveness of the vegetation to reduce leaching, surface runoff and fugitive dust. Plant production (peak standing crop) data will help guide the selection of potential plant species for waste stabilization. These data and plant metal levels will help guide the selection of post

Table 4.3.1. Vegetation seeded or sprigged at each STARS site.

Site	Seed Mix	Scientific Name Genus, Species, Cultivar	Common Name	Rate* (PLS/sq. meter)
2	1	<i>Agropyron elongatum</i> Orbit	Tall wheatgrass	215
		<i>Agropyron riparium</i> Sodar	Streambank wheatgrass	215
		<i>Agropyron intermedium</i> Greenar	Intermediate wheatgrass	215
		<i>Distichlis spicata</i>	Saltgrass	10 sprigs/plot
		<i>Lotus corniculatus</i> Tretana	Birdsfoot trefoil	215
		<i>Deschampsia caespitosa</i>	Tufted hairgrass	7 plants/plot
	2	<i>Agropyron trichophorum</i> Greenleaf	Pubescent wheatgrass	215
		<i>Agropyron trachycaulum</i> Revenue	Slender wheatgrass	215
		<i>Sitanion hystrix</i>	Bottlebrush squirreltail	215
		<i>Elymus junceus</i> Swift	Russian wildrye	215
		<i>Melilotus officinalis</i>	Yellow sweetclover	215
	7	<i>Festuca ovina</i> Covar	Sheep fescue	161
		<i>Poa compressa</i> Reubens	Canada bluegrass	161
		<i>Calamovilfa longifolia</i> Goshen	Prairie sandreed	108
		<i>Elymus augustus</i>	Altai wildrye	108
		<i>Agropyron dasystachyum</i> Critana	Thickspike wheatgrass	108
		<i>Melilotus officinalis</i>	Yellow sweetclover	108
	2	<i>Festuca longifolia</i> Durar	Hard sheep fescue	161
		<i>Deschampsia caespitosa</i>	Tufted hairgrass	161
		<i>Elymus giganteus</i> Volga	Mammoth wildrye	108
		<i>Elymus cinereus</i> Magnar	Basin wildrye	108
		<i>Agropyron dasystachyum</i> Critana	Thickspike wheatgrass	108
		<i>Lotus corniculatus</i> Tretana	Birdsfoot trefoil	108
21	1	<i>Agropyron elongatum</i> Orbit	Tall wheatgrass	161
		<i>Agropyron cristatum</i> Ephraim	Crested wheatgrass	161
		<i>Agropyron riparium</i> Sodar	Streambank wheatgrass	161
		<i>Distichlis spicata</i>	Saltgrass	10 sprigs/plot
		<i>Lotus corniculatus</i> Tretana	Birdsfoot trefoil	108
		<i>Deschampsia caespitosa</i>	Tufted hairgrass	7 plants/plot

Table 4.3.1. Continued.

Site	Seed Mix	Scientific Name Genus, Species, Cultivar	Common Name	Rate* (PLS/sq. meter)
	2	<i>Agropyron intermedium</i> Greenar	Intermediate wheatgrass	161
		<i>Agropyron desertorum</i> Nordan	Crested wheatgrass	161
		<i>Agropyron trichophorum</i> Greenleaf	Pubescent wheatgrass	161
		<i>Elymus junceus</i> Swift	Russian wildrye	161
		<i>Melilotus officinalis</i>	Yellow sweetclover	108
		<i>Artemisia longifolia</i>	Long leaf sagebrush	7 plants/plot
27	1	<i>Agropyron desertorum</i> Nordan	Crested wheatgrass	215
		<i>Agropyron smithii</i> Rosana	Western wheatgrass	161
		<i>Agropyron trichophorum</i> Greenleaf	Pubescent wheatgrass	161
		<i>Elymus augustus</i>	Altai wildrye	108
		<i>Lotus corniculatus</i> Tretana	Birdsfoot trefoil	108
	2	<i>Agropyron cristatum</i> Ephraim	Crested wheatgrass	215
		<i>Agropyron elongatum</i> Orbit	Tall wheatgrass	161
		<i>Agropyron intermedium</i> Greenar	Intermediate wheatgrass	161
		<i>Elymus junceus</i> Swift	Russian wildrye	108
		<i>Medicago sativa</i>	Alfalfa	108
33	1	<i>Agropyron smithii</i> Rosana	Western wheatgrass	161
		<i>Elymus cinereus</i> Magnar	Basin wildrye	108
		<i>Poa compressa</i> Reubens	Canada bluegrass	161
		<i>Agropyron trachycaulum</i> Revenue	Slender wheatgrass	108
		<i>Melilotus officinalis</i>	Yellow sweetclover	108
	2	<i>Elymus augustus</i>	Altai wildrye	108
		<i>Deschampsia caespitosa</i>	Tufted hairgrass	161
		<i>Agropyron trichophorum</i> Greenleaf	Pubescent wheatgrass	161
		<i>Agropyron riparium</i> Sodar	Streambank wheatgrass	108
		<i>Lotus corniculatus</i> Tretana	Birdsfoot trefoil	108

* PLS - Pure Live Seed.

rehabilitation land uses. Production data must be interpreted with care, however because it is strongly affected by the current year's climate. In addition, plant production on reclaimed lands often peaks between the third and sixth years.

Measurements of vegetation germination and establishment were made during the first growing season (see Appendix D1). Cover studies were conducted during 1989 (Appendix D2) and 1990 (Appendix D3).

In 1991, percent plant canopy coverage was estimated at peak growth using two 20 by 50 cm microplots within each seed mix in each treatment. Cover sampling was carried out in all replicated plots. Microplots were randomly located in areas representative of overall vegetation conditions in the study plot. Species specific plant canopy coverage was estimated using procedures outlined by Daubenmire (1959). The amount of plant litter, rock, gravel and bare ground, were also estimated (Appendix D4).

Production sampling was carried out in each replicate plot in the third growing season (1991). It was estimated by clipping all current year vegetation one centimeter above ground level in two 50 by 50 cm microplots within each seed mix in each plot. Microplots were randomly located in areas representative of overall vegetation conditions in the study plot. The clipped vegetation was separated by species and placed in paper bags. These samples were brought to the Reclamation Research laboratory, oven-dried at 70° C to a constant weight, and weighted (Appendix D5).

Plant cover and production at Site 2 (Manganese Stockpile) was not measured in 1991 because of the death of almost all plants on all STARS treatments at this site. Mortality of seeded vegetation was caused by high salinity exacerbated by periodic drought and poor drainage of the field site (Reclamation Research Unit 1992).

Rooting characteristics (Appendix D6) were determined by using a spade to expose a tailing/soil profile within each treatment combination of special interest. Profiles were excavated in areas where the vegetation was representative of that particular treatment. Within each exposed tailing/soil profile, maximum rooting depth (cm) relative to the zone of amendment incorporation was recorded.

Plant metal levels (Appendix D7) were determined by using clipped plant materials (see peak standing crop paragraph) from the dominant species within each seed mix-treatment combination. For each species, plant materials were composited from the eight subsample frames per seed mix. The material was dried, ground and determinations of total metals (Al, As, Cd, Cu, Fe, Mn, Pb, Hg, and Zn) were made.

To objectively designate superior plant species for the various wastes, "Importance Values" were generated for each dominant plant species at each site. Importance Values were calculated by determining a relative cover and a relative production value for each species and summing these values. This value ranks the species at a site among themselves

in terms of cover and production. It is not intended to be the basis of a comparison of plant performance between sites. The relative cover value was derived by dividing the individual plant cover by the sum of the cover of all species at the site and multiplying the quotient by 100. Relative production was derived from a similar expression utilizing production values.

Importance Values represent only the status of the vegetation at the time of sampling. It does not incorporate plant community succession into the expression. Therefore, hard sheep fescue and sheep fescue (*Festuca longifolia duriscula* and *Festuca ovina*) generate rather small Importance Values, but these numbers will continue to increase for several years. This is not represented in these calculated Importance Values. Slender wheatgrass (*Agropyron trachycaulum*) has a relatively strong Importance Value, but it will continue to decrease over time and this fact is not represented in these data.

4.3.1 Vegetation Response for STARS Site 7 - Rocker

Table 4.3.2 summarizes the 1991 plant cover data for the Rocker Study Site. Among the STARS treatments at Site 7, the Deep Plow treatment produced significantly greater total plant cover than the Controls in Seed Mix 1. This treatment also produced significantly greater total plant cover than the Control and the Slurry Injection in Seed Mix 2. Grass or forb cover alone did not reveal any significant differences between any of the STARS treatments and the Controls or the Topsoil Wedge. This was due to the inherent variability in reclaimed grassland cover values and the small number of samples.

The production data (Table 4.3.3) revealed numerous significant differences among the various treatments in grass and total production. The most numerous differences were between the Topsoil Wedge and the Control or between the STARS Deep Plow treatments and the Control. On the basis of plant cover and production data at this site, the STARS Deep Plow treatment must be designated as the most desirable soil manipulation attempted at Site 7.

Importance Values (Table 4.3.4) were not calculated for nonseeded weedy species nor were they determined for the Control plots. Control plots generally were devoid of vegetation and data from them only reinforces the plant inhibitory information previously generated for these materials. Two of the wildryes performed well in this sandy soil. They were established and reproducing within three growing seasons. Mammoth wildrye (*Elymus giganteus*) established well and was spreading by rhizomes. Altai wildrye (*Elymus augustus*) was also very aggressive. Thickspike wheat grass (*Agropyron dasystachyum*) was spreading vegetatively on the plots. Basin wildrye (*Elymus cinereus*) performed better on the topsoiled plots than on the STARS plots. Legumes did not perform well on this site, but they can be expected to increase in cover and production throughout the next few growing seasons.

Table 4.3.2. Mean vegetation cover (% areal cover) at STARS Site 7 (Rocker), Summer 1991.

Treatment	Seed Mix 1			Seed Mix 2			Mean of Total Cover
	Grasses	Forbs	Total	Grasses	Forbs	Total	
Agricultural	22.2 A ¹	5.9 A	28.1 AB	35.4 A	6.0 A	41.3 BC	34.8 B
Control	0.5 A	0.0 A	0.5 A	0.5 A	<0.1 A	0.5 A	2.6 A
Deep Plow	41.9 A	3.5 A	45.3 B	47.9 A	4.3 A	52.2 C	48.8 CB
Slurry Injection	28.3 A	2.8 A	31.1 AB	20.2 A	<0.1 A	20.2 AB	25.7 AB
Topsoil Wedge							
38 cm	33.4 A	30.4 A	62.6 CB	41.1 A	29.5 A	70.6 C	66.6 CD
15 cm	49.5 A	44.3 A	93.0 C	36.5 A	32.9 A	69.4 C	81.2 D

¹ Multiple mean comparison based on LSD at significance level of 0.10. Mean followed by same letter in *columns* are not different.

Table 4.3.3. Mean vegetation production (kg/ha¹) at STARS Site 7 (Rocker), Summer 1991.

Treatment	Seed Mix 1			Seed Mix 2			Mean of Total Production
	Grasses	Forbs	Total	Grasses	Forbs	Total	
Agricultural	528 AB ²	30.2 A	558 AB	829 BC	29.0 A	858 AB	708 AB
Control	31 A	0.0 A	31 A	8.5 A	0.5 A	8.9 A	19.6 A
Deep Plow	1530 B	18.0 A	1548 B	1211 C	47.3 A	1258 BC	1403 BC
Slurry Injection	531 AB	4.1 A	535 AB	171 AB	0.5 A	172 A	353 A
Topsoil Wedge	1242 AB	498 B	1740 B	792 BC	1159 B	1950 C	1845 C

¹ To convert to lbs/a multiply by 0.89.

² Multiple mean comparison based on LSD at significance level of 0.10. Mean followed by same letter in columns are not different.

Table 4.3.4. Importance values for the dominant seeded species on Site 7 (Rocker), Summer 1991.

Species (Seed Mix 1)	Relative % Cover	Relative % Production	Importance Value ¹	Species (Seed Mix 2)	Relative % Cover	Relative % Production	Importance Value ¹
<u>Agricultural</u>				<u>Agricultural</u>			
Ag da ²	18.7	23.7	42.4	Ag da	9.5	9.7	19.2
El an	29.9	38.0	67.9	El ci	16.3	22.8	39.1
Fe ov	10.3	11.0	21.3	El gi	58.5	48.8	107.3
Po co	1.6	4.4	6.0	Fe lo	1.1	2.6	3.7
Legumes	17.1	1.4	18.5	Legumes	11.7	1.5	13.2
<u>Deep Plow</u>				<u>Deep Plow</u>			
Ag da	13.8	19.5	33.3	Ag da	11.6	23.8	35.4
El an	59.5	74.7	134.2	El ci	3.7	12.4	16.1
Fe ov	10.5	1.8	12.3	El gi	62.7	57.5	120.2
Po co	0.0	0.0	0.0	Fe lo	13.7	2.1	15.8
Legumes	7.6	1.2	8.8	Legumes	0.0	3.6	3.6
<u>Slurry Injection</u>				<u>Slurry Injection</u>			
Ag da	32.2	31.7	63.9	Ag da	13.9	10.3	24.2
El an	48.3	59.1	107.4	El ci	0.3	6.5	6.8
Fe ov	0.0	0.0	0.0	El gi	85.4	82.8	168.2
Po co	0.0	1.1	1.1	Fe lo	0.0	0.0	0.0
Legumes	7.7	0.2	7.9	Legumes	0.0	0.0	0.0
<u>Topsoil Wedge</u>				<u>Topsoil Wedge</u>			
Ag da	14.8	27.7	42.5	Ag da	12.1	13.2	25.3
El an	25.8	35.7	61.5	El ci	28.3	24.8	53.7
Fe ov	11.7	7.5	19.1	El gi	3.2	0.1	3.3
Po co	0.0	0.0	0.0	Fe lo	6.9	2.4	9.3
Legumes	23.0	3.8	26.8	Legumes	15.9	6.6	22.5

¹ Importance Value = Relative % Cover + Relative % Production.

² See Appendix D8 for vegetation names and symbols.

The fescues (*Festuca ovina* and *Festuca longifolia*) can also be expected to increase in cover if the other grasses do not close the canopy over them. The fescues need direct sun and continued rapid growth of rhizomatous species will shade them out of the plant community.

For the type of waste material at Site 7 and for the method of contaminant immobilization, Mammoth wildrye, (*Elymus giganteus*), Altai wildrye (*Elymus augustus*), and Thicksipike wheatgrass (*Agropyron dasystachyum*) constitute an aggressive seed mix. A legume is essential to the development of a functioning ecosystem and should be added to such a mix if it is seeded.

Metal levels in and on two of the dominant grasses at this site are shown in Table 4.3.5. For comparison, metal analyses of related wheatgrasses (*Agropyron smithii*) collected in southeastern Montana are presented in this table. Although elevated, the metal levels in and on the vegetation from this site will not pose a threat to grazing wildlife or the occasional livestock that might pass through this area along Silver Bow Creek. The National

Table 4.3.5. Comparison of mean elemental levels (mg/kg) in dominant vegetation collected (July 1991) from STARS Site 7 (Rocker).

Element	Agricultural Tillage	Control	Treatment			SE Montana Western Wheatgrass ² (<i>Agropyron Smithii</i>)
			Deep Plow	Slurry Injection	Topsoil	
Thicksipike wheatgrass (<i>Agropyron dasystachyum</i>)						
Aluminum	90.2 B ¹	119.0 CB	86.1 B	165.0 C	33.4 A	5 to 50
Arsenic	3.1 A	2.2 A	2.1 A	2.5 A	3.9 A	<1 to 2
Cadmium	0.27 B	0.41 B	0.29 B	0.23 AB	0.12 A	0.01 to 0.1
Copper	14.9 B	26.6 C	15.7 CB	21.7 CB	6.4 A	2.5 to 4.5
Lead	12.3 A	3.90 A	8.8 A	6.4 A	20.9 A	1.0 to 2.0
Manganese	78.1 A	58.4 A	53.3 A	76.4 A	59.4 A	25 to 65
Zinc	69.7 A	72.0 A	56.4 A	47.1 A	35.6 A	12 to 21
Altai wildrye (<i>Elymus augustus</i>)						
Aluminum	66.5 AB	³	62.9 AB	134.0 B	36.4 A	
Arsenic	7.1 A		3.3 A	7.6 A	1.8 A	
Cadmium	0.39 A		0.36 A	0.26 A	0.09 A	
Copper	13.4 A		12.2 A	15.0 A	6.0 A	
Lead	14.8 A		3.3 A	6.7 A	1.2 A	
Manganese	75.1 A		53.0 A	60.8 A	71.0 A	
Zinc	63.8 A		57.2 A	44.4 A	42.4 A	

¹ Multiple mean comparison based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

² Al and As estimated from the literature, Cd, Cu, Pb, Mn, and Zn from Munshower et al. 1987.

³ Species did not grow on control plot.

Research Council has established mineral tolerances for domestic animals (NRC 1980). Maximum tolerable levels (mg/kg) of dietary elements for cattle are: aluminum (1000), inorganic arsenic (50), cadmium (0.5), copper (100), lead (30), Manganese (1000), and zinc (500). These levels are defined by the NRC as: "... when fed for a limited period will not impair animal performance and should not produce unsafe residues in human food derived from the animal".

Two soil pits were opened at Site 7 to examine root penetration and distribution. All soil pits were placed in Deep Plowed treatments because previous pits had provided evidence that root penetration was deepest in this treatment. In both pits, roots penetrated to 25 cm (10 inches). There did not appear to be any impediment to deeper penetration and the plant roots may still be developing. It is doubtful that the roots have reached their point of maximum penetration.

4.3.2 Vegetation Response for STARS Site 21 - Ramsay Flats

Table 4.3.6 summarizes plant cover measured in 1991 at STARS Site 21 (Ramsay Flats). Both grass and total cover data revealed significant differences among the treatments. Cover in these two categories was higher on all treatments than on the Control plots. As at Site 7, the average total cover on the Agricultural and the Deep Plow treatments were similar. At this site, as at Site 7, the Topsoil Wedge usually did not reveal higher cover values than the Deep Plow treatment.

The production data (Table 4.3.7) also revealed differences between the Control plots and the STARS treatments, but there were almost no differences between the STARS treatments and the Topsoil Wedge. As at Site 7, production and cover on the Deep Plow plots appeared to be very successful and on this site it was comparable to these parameters on the Topsoil Wedge.

Importance Values suggest that the crested wheatgrasses performed very well on the Topsoil Wedges but not on the STARS treatments (Table 4.3.8). On STARS Seed Mix 1 plots, Tall wheatgrass (*Agropyron elongatum*) dominated the landscape. Intermediate wheatgrass (*Agropyron intermedium*) revealed an identical pattern on Seed Mix 2 plots. Both of these wheatgrasses shared dominance with the crested wheatgrass species on the Topsoil Wedges.

Metal levels in or on the plant tissues from the STARS plots at Site 21 (Table 4.3.9) were generally ten times those found in similar vegetation at Site 7. The differences between metal levels in the vegetation growing on the Topsoil Wedges at the two sites was much less. Lower metal levels in this vegetation may be due to lower metal levels in the root zone or reduced dust accumulation on the elevated wedges.

At this site, the possibility of trace element imbalances in livestock and wildlife forages must be considered. Copper, cadmium, lead, and zinc levels in and on vegetation are high enough to produce problems to animals grazing these sites throughout the year.

Table 4.3.6. Mean vegetation cover (% areal cover) at STARS Site 21 (Ramsay Flats), Summer 1991.

Treatment	Seed Mix 1			Seed Mix 2			Mean of Total Cover
	Grasses	Forbs	Total	Grasses	Forbs	Total	
Agricultural	56.8 B ¹	0.5 A	57.3 CB	53.5 CB	0.0 A	53.5 CB	55.4 CB
Control	0.0 A	0.0 A	0.0 A	0.0 A	0.0 A	0.0 A	0.0 A
Deep Plow	55.0 B	7.3 A	62.3 CB	59.7 C	2.0 A	61.7 C	62.0 C
Slurry Injection	47.7 B	2.4 A	50.1 B	38.4 B	0.6 A	39.0 B	44.6 B
Topsoil Wedge							
38 cm	65.3 CB	4.4 A	69.7 CD	63.4 C	0.3 A	63.6 C	66.7 CD
15 cm	81.6 C	3.9 A	85.5 D	67.8 C	2.4 A	70.1 C	77.8 D

¹ Multiple mean comparison based on LSD at significance level of 0.10. Mean followed by same letter in *columns* are not different.

Table 4.3.7. Mean vegetation production (kg/ha¹) at STARS Site 21 (Ramsay Flats), Summer 1991.

Treatment	Seed Mix 1			Seed Mix 2			Mean of Total Production
	Grasses	Forbs	Total	Grasses	Forbs	Total	
Agricultural	1883 B ²	2.6 A	1885 B	2065 B	3.1 A	2068 B	1977 CB
Control	0 A	0.0 A	0 A	0 A	0.0 A	0 A	0 A
Deep Plow	2805 B	126 B	2932 B	2354 B	17.4 A	2371 B	2651 C
Slurry Injection	2128 B	21.3 A	2149 B	1299 AB	1.8 A	1301 AB	1725 B
Topsoil Wedge	2274 B	27.0 A	2301 B	2006 B	6.7 A	2017 B	2159 CB

¹ To convert to lbs/a multiply by 0.89.

² Multiple mean comparison based on LSD at significance level of 0.10. Mean followed by same letter in *columns* are not different.

Table 4.3.8. Importance values for the dominant seeded species on Site 21 (Ramsay Flats), Summer 1991.

Species (Seed Mix 1)	Relative % Cover	Relative % Production	Importance Value ¹	Species (Seed Mix 2)	Relative % Cover	Relative % Production	Importance Value ¹
<u>Agricultural</u>				<u>Agricultural</u>			
Ag cr ²	0.0	0.0	0.0	Ag de	1.6	1.0	7.6
Ag el	98.3	98.5	196.8	Ag in	74.2	63.2	137.4
Ag ri	0.0	0.4	0.4	Ag tri	3.6	0.0	3.6
Legumes	0.0	0.0	0.0	El ju	16.9	35.3	52.2
				Legumes	0.0	0.0	0.0
<u>Deep Plow</u>				<u>Deep Plow</u>			
Ag cr	3.8	3.0	6.8	Ag de	3.9	3.6	7.5
Ag el	80.7	89.6	170.3	Ag in	64.0	67.7	131.7
Ag ri	0.0	0.3	0.3	Ag tri	0.7	3.0	3.7
Legumes	0.7	0.0	0.7	El ju	25.1	19.7	44.8
				Legumes	0.0	0.0	0.0
<u>Slurry Injection</u>				<u>Slurry Injection</u>			
Ag cr	0.0	0.7	0.7	Ag de	7.2	1.8	9.0
Ag el	94.3	93.2	187.5	Ag in	45.4	61.5	106.9
Ag ri	0.1	0.7	0.8	Ag tri	0.0	0.0	0.0
Legumes	0.0	0.0	0.0	El ju	44.8	36.6	81.4
				Legumes	0.0	0.0	0.0
<u>Topsoil Wedge</u>				<u>Topsoil Wedge</u>			
Ag cr	45.8	48.9	94.7	Ag de	46.3	46.2	92.5
Ag el	45.2	47.2	92.4	Ag in	30.9	38.6	69.5
Ag ri	3.7	2.7	6.4	Ag tri	10.1	12.1	22.2
Legumes	0.0	0.0	0.0	El ju	10.0	0.0	10.0
				Legumes	0.0	0.0	0.0

¹ Importance Value = Relative % Cover + Relative % Production.

² See Appendix D8 for vegetation names and symbols.

Table 4.3.9. Comparison of mean elemental levels (mg/kg) in dominant vegetation collected (July 1991) from STARS Site 21 (Ramsay Flats).

Element	Agricultural Tillage	Control	Treatment			SE Montana Western Wheatgrass ² (<i>Agropyron Smithii</i>)
			Deep Plow	Slurry Injection	Topsoil	
Tall wheatgrass (<i>Agropyron elongatum</i>)						
Aluminum	316.0 AB ¹	³	384.0 B	302.0 AB	74.0 A	5 to 50
Arsenic	26.1 AB		33.2 B	21.9 AB	4.3 A	<1 to 2
Cadmium	2.0 B		1.7 AB	1.5 AB	0.61 A	0.01 to 0.1
Copper	278.0 B		247.0 AB	216.0 AB	63.9 A	2.5 to 4.5
Lead	30.5 AB		60.6 B	25.1 AB	5.4 A	1.0 to 2.0
Manganese	917.0 B		672.0 AB	512.0 A	320.0 A	25 to 65
Zinc	750.0 B		541.0 AB	393.0 A	249.0 A	12 to 21
Intermediate wheatgrass (<i>Agropyron intermedium</i>)						
Aluminum	311.0 B	³	266.0 B	286.0 B	48.4 A	
Arsenic	21.7 B		22.3 B	26.6 B	3.1 A	
Cadmium	1.9 B		1.6 AB	2.1 B	0.57 A	
Copper	228.0 B		196.0 AB	268.0 B	59.5 A	
Lead	28.3 B		38.2 B	29.7 B	3.5 A	
Manganese	790.0 B		622.0 B	677.0 B	243.0 A	
Zinc	589.0 B		491.0 AB	575.0 B	195.0 A	

¹ Multiple mean comparison based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

² Al and As estimated from the literature, Cd, Cu, Pb, Mn, and Zn from Munshower et al. 1987).

³ Species did not grow on control plot.

These levels are elevated above levels recommended for livestock consumption (NRC 1980). It is thought that these high metal and arsenic loads are predominantly *on* the vegetation rather than *in* the vegetation. The source of the metals/arsenic is most probably a result of dust and raindrop splash. The experimental plots at this site are surrounded by approximately 65 hectares (160 acres) of tailings and wastes devoid of vegetation and tailings movement via wind is common. During the Phase I greenhouse portion of this work, metal levels (except Cd) *in* vegetation were much lower than those exhibited for field grown plants (Reclamation Research Unit, Schafer & Associates, and CH2M HILL, Inc. 1989b). Loadings of the metals (Cu, Mn, and Zn) are also higher for plants grown on the coversoil wedge at this site compared to loadings for plants grown on the wedges at the other sites. We may infer, therefore, that the very high metal levels in field grown plants are a result of surface contamination rather than internal (tissue) metal loads.

Root penetration at Site 21 was quite variable. In two soil pits opened in one plot amended by slurry injection, roots penetrated to 12 cm (5 inches) and 20 cm (8 inches). In a replicate of the slurry injection treatment, root penetration was to 17.5 cm (7 inches). In a third replication of this treatment, roots had penetrated to 27.5 cm (11 inches). This latter

example was noted as the best revegetated slurry injection plot. In the only Agricultural plot examined, roots stopped growing down and branched laterally at 17.5 cm (7 inches). This was the limit of lime incorporation on this plot. In a Deep Plow treatment at Site 21, roots had penetrated to 38 cm (15 inches). Even at this depth there was no indication that the roots were at their limit of penetration. They appeared to be growing deeper.

The increase in production (and cover) of the crested wheatgrasses (*Agropyron cristatum* and *Agropyron desertorum*) on the Topsoil Wedges indicates that these species will do well when a coversoil is available, but are not capable of successfully competing on finer textured, acid producing wastes even if amended.

On this type of waste material, direct seeding into the amended tailings of Tall wheatgrass, Intermediate wheatgrass, and Russian wildrye (*Elymus junceus*) appear to be most successful. Their use with a legume must be recommended.

4.3.3 Vegetation Response for STARS Site 27 - Fairmont

Table 4.3.10 is a summary of plant cover at this STARS site. Since this site consisted of a contaminated soil rather than wastes, no Topsoil Wedge was constructed there. There were significant differences between production on the Control plots and the treatments. Seed Mix 1 also showed differences in total production between the Extra treatment and the Agricultural or Deep Plow treatments. Seed Mix 2 did not reveal any significant difference other than those between the Controls and the treatments. The Agricultural and Deep Plow treatments generally revealed equal cover measurements.

Production values (Table 4.3.11) reflected trends exhibited in the measurement of cover. Higher forb production on some of these treatments reflects the presence of Rushes (*Juncus balticus*) on this site and their increase and persistence after treatment.

Both of the introduced crested wheatgrasses performed very well on this site, but neither Altai nor Russian wildrye had high Importance Values (Table 4.3.12). Since this site was in a pasture, the crested wheatgrasses were very appropriate for the designated land use. Because of the performance of the crested wheatgrasses on the Topsoil Wedges at Site 21, the performance of these species in and around Butte, Montana (Keammerer et al. 1992) and their performance at this site, these species should be restricted to use on coversoiled sites. Tufted hairgrass (*Deschampsia caespitosa*) was not seeded at this site yet it invaded the site and revealed an average Importance Value of 6.7.

Higher level of phosphorus fertilization used on the Extra plots resulted in lower plant cover and production values compared to vegetation grown on the plots treated with agricultural tillage and nominal levels of phosphorus. These results are different from those obtained in the greenhouse where additional phosphorus improved plant performance. Causes for differences in field and greenhouse plant performance are unknown.

Table 4.3.10. Mean vegetation cover (% areal cover) at STARS Site 27 (Fairmont), Summer 1991.

Treatment	Seed Mix 1			Seed Mix 2			Mean of Total Cover
	Grasses	Forbs	Total	Grasses	Forbs	Total	
Agricultural	70.2 C ¹	1.4 A	71.6 C	75.5 B	1.9 A	77.3 B	74.5 C
Control	29.0 A	2.0 A	31.0 A	4.5 A	0.0 A	4.5 A	17.8 A
Deep Plow	60.7 BC	2.2 A	62.9 C	62.2 B	1.2 A	63.3 B	63.2 CB
Extra ²	41.8 AB	3.4 A	45.1 AB	68.5 B	3.5 A	72.0 B	58.6 B

¹ Multiple mean comparison based on LSD at significance level of 0.10. Mean followed by same letter in *columns* are not different.

² This treatment consisted of an agricultural treatment, but extra phosphorus was added.

Table 4.3.11. Mean vegetation production (kg/ha¹) at STARS Site 27 (Fairmont), Summer 1991.

Treatment	Seed Mix 1			Seed Mix 2			Mean of Total Production
	Grasses	Forbs	Total	Grasses	Forbs	Total	
Agricultural	3827 C ²	37.4 AB	3865 C	3282 B	61.3 A	3344 B	3604 C
Control	128 A	81.4 B	209 A	75.7 A	24.3 A	100 A	155 A
Deep Plow	4249 C	5.6 A	4254 C	3087 B	12.7 A	3099 B	3677 C
Extra	2106 B	16.6 AB	2123 B	2893 B	41.7 A	2939 B	2531 B

¹ To convert to lbs/a multiply by 0.89.

² Multiple mean comparison based on LSD at significance level of 0.10. Mean followed by same letter in *columns* are not different.

Table 4.3.12. Importance values for the dominant seeded species on Site 27 (Agricultural), Summer 1991.

Species (Seed Mix 1)	Relative % Cover	Relative % Production	Importance Value ¹	Species (Seed Mix 2)	Relative % Cover	Relative % Production	Importance Value ¹
<u>Agricultural</u>				<u>Agricultural</u>			
Ag de ²	61.7	82.6	144.3	Ag cr	46.7	50.0	96.7
Ag sm	5.4	1.0	6.4	Ag el	1.1	14.8	15.9
Ag tri	10.8	7.0	17.8	Ag in	6.7	3.1	9.8
El an	6.6	5.5	12.1	El ju	43.5	29.2	73.7
Legumes	0.0	0.0	0.0	Legumes	0.6	0.2	0.8
<u>Deep Plow</u>				<u>Deep Plow</u>			
Ag de	89.1	91.0	180.1	Ag cr	63.0	69.7	132.7
Ag sm	2.1	0.3	2.4	Ag el	1.4	9.1	10.5
Ag tri	1.5	3.9	5.4	Ag in	8.2	1.9	10.1
El an	3.8	4.6	8.4	El ju	24.9	18.9	43.8
Legumes	0.0	0.0	0.0	Legumes	0.8	0.1	0.9
<u>Extra</u>				<u>Extra</u>			
Ag de	36.3	74.0	110.3	Ag cr	24.3	50.1	74.4
Ag sm	11.7	2.6	14.3	Ag el	5.4	6.1	11.5
Ag tri	11.5	7.0	18.5	Ag in	0.7	2.0	2.7
El an	5.3	6.6	12.9	El ju	61.4	39.4	100.8
Legumes	0.0	0.0	0.0	Legumes	0.0	0.1	0.1

¹ Importance Value = Relative % Cover + Relative % Production.

² See Appendix D8 for vegetation names and symbols.

Metal levels (Table 4.3.13) were lower in and on these plants than the levels found in or on vegetation at Sites 7 and 21. Only cadmium levels were high enough to pose a potential problem to livestock continuously feeding at the site.

Root penetration in soils at Site 27 was quite variable. In the soils receiving the Agricultural treatment, root penetration appeared to stop at the depth of incorporation or approximately 17.5 cm (7 inches). In two plots receiving the Extra treatment, plant

Table 4.3.13. Comparison of mean elemental levels (mg/kg) in dominant vegetation collected (July 1991) from STARS Site 27 (Fairmont).

Element	Treatment				SE Montana Western Wheatgrass ¹ (<i>Agropyron Smithii</i>)
	Agricultural Tillage	Control	Deep Plow	Extra	
Crested wheatgrass (<i>Agropyron cristatum</i>)					
Aluminum	99.0 A ¹	³	49.6 A	77.4 A	5 to 50
Arsenic	3.4 A		1.7 A	3.3 A	<1 to 2
Cadmium	0.58 A		0.54 A	0.35 A	0.01 to 0.1
Copper	18.6 A		10.3 A	20.0 A	2.5 to 4.5
Lead	11.8 A		6.2 A	12.1 A	1.0 to 2.0
Manganese	75.1 A		57.9 A	75.4 A	25 to 65
Zinc	119.0 A		122.0 A	111.0 A	12 to 21
Desert wheatgrass (<i>Agropyron desertorum</i>)					
Aluminum	47.6 A	³	61.3 A	101.0 A	
Arsenic	2.1 A		2.1 A	4.9 A	
Cadmium	0.45 A		0.53 A	0.60 A	
Copper	10.6 A		12.0 A	20.4 A	
Lead	6.3 A		8.0 A	21.9 A	
Manganese	59.4 A		52.3 A	81.6 A	
Zinc	126.0 A		124.0 A	160.0 A	
Russian wildrye (<i>Elymus junceus</i>)					
Aluminum	82.5 A	³	42.4 A	67.0 A	
Arsenic	3.1 A		2.6 A	3.8 A	
Cadmium	1.1 A		0.84 A	0.60 A	
Copper	19.1 A		11.7 A	20.9 A	
Lead	15.5 A		7.2 A	11.5 A	
Manganese	89.9 A		69.6 A	75.8 A	
Zinc	154.0 A		115.0 A	121.0 A	

¹ Multiple mean comparison based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

² Al and As estimated from the literature, Cd, Cu, Pb, Mn, and Zn from Munshower et al. 1987).

³ Species did not grow on control plot.

roots extended to a sand layer at 15 cm (6 inches) in one plot and to 20 cm (8 inches) in the other. In the first plot, the roots appeared to have ceased penetration and were developing laterally. In the second plot, the roots were still growing down and did not appear to have

reached an impediment to their growth. In a soil pit opened in a Deep Plow treatment, roots were found at 30 cm (12 inches) and they did not appear to have reached an inhibitory zone.

Plant species that exhibited good performance and can be recommended for similar areas within the Operable Unit are Crested and Desert wheatgrasses and Russian wildrye. All three withstand grazing and are desirable forage species.

4.3.4 Vegetation Response for STARS Site 33 - Opportunity

Table 4.3.14 contains a summary of vegetation cover values from STARS Site 33. These data reveal few significant differences among the various treatments. The Control plots always revealed less grass and total cover than the treatments and the Topsoil Wedge, Seed Mix 1, revealed greater grass and total cover at 38 cm soil depth than most of the STARS treatments. The shrub cover was composed entirely of greasewood (*Sarcobatus vermiculatus*). It was volunteering from seed and shallow roots sprouting in the treated soils. There was no greasewood on the Topsoil Wedges because the seed and roots had been buried under several centimeters of coversoil.

Plant production (Table 4.3.15) revealed trends similar to those found in the cover data. Production on the Deep Plow, Slurry Injection, and Topsoil Wedge was usually similar and greater than production on the Agricultural treatment and the Control.

The Importance Values (Table 4.3.16) for all of the wheatgrasses (*Agropyron* spp.) were rather low on this site. The wildryes, especially Altai wildrye, performed well in this saline soil. The wildryes generally perform better in saline soils than the wheatgrasses.

Saltgrass (*Distichlis spicata*) had a mean Importance Value of 22.3 at this site even though the species was not seeded. All of this plant cover and production resulted from rhizome growth in these soils. While there were scattered saltgrass plants on this site before amendment, post-treatment growth of the species appears to have been stimulated by the amendments.

Metal levels in vegetation from this site (Table 4.3.17) were lower than those from any other STARS sites. The levels in plants growing on the Topsoil Wedges were, for several elements, within the ranges found in nonpolluted environments. Cadmium levels, however, were near the maximum tolerable level for livestock consumption.

One soil pit was opened in an Agricultural treatment at Site 33. In this plot, roots had penetrated to 20 cm (8 inches) and had then grown laterally. In soils amended by slurry injection, the vast majority of the roots were found in the 15 to 20 cm (6 to 8 inch)

Table 4.3.14. Mean vegetation cover (% areal cover) at STARS Site 33 (Opportunity), Summer 1991.

Treatment	Seed Mix 1				Seed Mix 2				Mean of Total Cover
	Grasses	Forbs	Shrubs	Total	Grasses	Forbs	Shrubs	Total	
Agricultural	35.9 B ¹	8.2 A	14.9 A	58.9 B	46.2 B	4.8 A	2.2 AB	53.0 B	56.0 B
Control	11.7 A	0.0 A	0.0 A	11.7 A	12.9 A	0.0 A	0.0 A	13.0 A	12.3 A
Deep Plow	41.1 B	17.3 A	15.6 A	74.0 CB	58.3 B	4.3 A	11.5 BC	74.2 B	74.0 CB
Slurry Injection	42.5 B	5.5 A	9.5 A	57.5 B	60.0 B	0.5 A	13.4 C	73.8 B	65.7 CB
Topsoil Wedge									
38 cm	69.5 C	23.9 A	0.0 A	93.4 C	56.1 CB	3.0 A	0.0 A	59.1 B	76.3 C
15 cm	45.9 B	30.3 A	0.0 A	76.1 CB	49.6 CB	2.9 A	0.0 A	52.5 B	64.3 CB

¹ Multiple mean comparison based on LSD at significance level of 0.10. Mean followed by same letter in *columns* are not different.

Table 4.3.15. Mean vegetation production (kg/ha¹) at STARS Site 33 (Opportunity), Summer 1991.

Treatment	Seed Mix 1				Seed Mix 2				Mean of Total Cover
	Grasses	Forbs	Shrubs	Total	Grasses	Forbs	Shrubs	Total	
Agricultural	819 A ²	83.2 AB	266 A	1169 B	984 A	72.0 BC	168 A	1224 A	1196 AB
Control	276 A	0.0 A	0 A	276 A	365 A	0.0 A	0 A	365 A	320 A
Deep Plow	1217 AB	178 AB	634 A	2029 C	3362 B	0.0 A	359 A	3725 B	2877 C
Slurry Injection	2175 C	49.1 AB	420 A	2645 C	1515 AB	21.9 AB	350 A	1887 AB	2266 C
Topsoil Wedge	1843 BC	304 B	0 A	2148 C	2141 AB	85.0 C	0 A	2227 AB	2187 BC

¹ To convert to lbs/a multiply by 0.89.

² Multiple mean comparison based on LSD at significance level of 0.10. Mean followed by same letter in *columns* are not different.

Table 4.3.16. Importance values for the dominant seeded species on Site 33 (Opportunity), Summer 1991.

Species (Seed Mix 1)	Relative % Cover	Relative % Production	Importance Value ¹	Species (Seed Mix 2)	Relative % Cover	Relative % Production	Importance Value ¹
<u>Agricultural</u>				<u>Agricultural</u>			
Ag sm ²	1.7	4.3	6.0	Ag ri	22.3	9.7	31.0
Ag tra	6.6	3.7	10.3	Ag tri	23.3	9.8	33.1
El ci	3.4	8.5	11.9	El an	32.6	35.2	67.8
Po co	0.0	0.1	0.1	De ca	0.8	0.1	0.9
Legumes	1.7	1.8	3.5	Legumes	0.0	0.0	0.0
<u>Deep Plow</u>				<u>Deep Plow</u>			
Ag sm	7.9	2.9	10.8	Ag ri	0.8	1.8	2.6
Ag tra	0.1	20.7	20.8	Ag tri	9.0	5.2	14.2
El ci	33.3	24.3	57.6	El an	65.8	80.6	146.4
Po co	0.0	0.0	0.0	De ca	2.6	0.0	2.6
Legumes	20.1	8.4	28.5	Legumes	2.6	0.0	2.6
<u>Slurry Injection</u>				<u>Slurry Injection</u>			
Ag sm	8.4	7.5	15.9	Ag ri	4.4	26.0	30.4
Ag tra	4.1	18.5	22.6	Ag tri	12.3	3.7	16.0
El ci	38.2	47.5	85.8	El an	51.7	45.2	96.9
Po co	0.1	0.0	0.1	De ca	0.7	0.0	0.7
Legumes	3.6	0.1	3.7	Legumes	0.0	0.0	0.0
<u>Topsoil Wedge</u>				<u>Topsoil Wedge</u>			
Ag sm	0.0	0.8	0.8	Ag ri	10.1	8.5	18.6
Ag tra	18.1	13.0	31.1	Ag tri	33.6	20.9	54.5
El ci	47.1	71.1	118.2	El an	44.1	52.3	96.4
Po co	0.0	0.0	0.0	De ca	3.5	0.2	3.7
Legumes	7.5	1.2	8.7	Legumes	1.9	0.2	2.1

¹ Importance Value = Relative % Cover + Relative % Production.

² See Appendix D8 for vegetation names and symbols.

Table 4.3.17. Comparison of mean elemental levels (mg/kg) in dominant vegetation collected (July 1991) from STARS Site 33 (Opportunity).

Element	Treatment				SE Montana Western Wheatgrass ³ (<i>Agropyron Smithii</i>)	
	Agricultural Tillage	Control	Deep Plow	Slurry Injection		Topsoil
Streambank wheatgrass (<i>Agropyron riparium</i>)						
Aluminum	95.2 A ¹	³	97.1 A	177.0 A	70.6 A	5 to 50
Arsenic	3.3 A		3.1 A	4.0 A	1.8 A	<1 to 2
Cadmium	0.54 BC		0.68 C	0.30 AB	0.13 A	0.01 to 0.1
Copper	35.2 AB		31.0 AB	50.3 B	14.3 A	2.5 to 4.5
Lead	4.5 A		4.3 A	6.6 A	1.4 A	1.0 to 2.0
Manganese	217.0 B		141.0 A	140.0 A	95.0 A	25 to 65
Zinc	84.1 B		103.0 B	74.2 AB	42.2 A	12 to 21
Altai wildrye (<i>Elymus augustus</i>)						
Aluminum	153.0 A	³	129.0 A	196.0 A	76.3 A	
Arsenic	3.2 AB		2.3 AB	3.7 B	1.6 A	
Cadmium	0.98 B		0.76 AB	0.49 AB	0.32 A	
Copper	46.8 B		25.5 AB	45.1 B	15.2 A	
Lead	5.5 A		3.5 A	5.3 A	1.3 A	
Manganese	270.0 B		107.0 A	166.0 A	133.0 A	
Zinc	236.0 C		177.0 BC	126.0 AB	90.3 A	
Basin wildrye (<i>Elymus cinereus</i>)						
Aluminum	57.9 A	³	65.9 A	92.5 A	83.3 A	
Arsenic	2.4 A		1.9 A	1.7 A	2.1 A	
Cadmium	0.45 AB		0.61 B	0.38 AB	0.20 A	
Copper	21.8 A		21.7 A	28.0 A	16.5 A	
Lead	2.0 A		2.0 A	2.4 A	1.3 A	
Manganese	128.0 A		111.0 A	144.0 A	98.5 A	
Zinc	112.0 A		112.0 A	98.2 A	63.8 A	

¹ Multiple mean comparison based on LSD at significance level of 0.05. Means followed by same letter in rows are not different.

² Al and As estimated from the literature, Cd, Cu, Pb, Mn, and Zn from Munshower et al. 1987).

³ Species did not grow on control plot.

zone, but saltgrass rhizomes were found at 35 cm (14 inches). Two pits were opened in soils amended by the Deep Plow. In both plots, the root mass extended to 25 cm (10 inches) but individual roots had penetrated to 38 cm (15 inches) and appeared to be continuing their vertical growth at this depth. One soil pit was opened in the middle of a Topsoil Wedge at this site. Roots had developed into the underlying amended material, but had not grown deeper than this layer. Saltgrass rhizomes were observed, however, in materials below the amended soil under the wedge.

Two species, Altai wildrye and Basin wildrye, performed well on the amended wastes at this site.

4.3.5 Vegetation Response Summary

Plant performance on the STARS plots was inferred from measurements of species areal cover, species production, plant rooting depth, and plant species metal levels. At almost every site, vegetation production and cover on the Control plots was significantly lower than the same parameters on the amended plots. When these two parameters were measured on the Topsoil Wedge they were usually comparable to these same parameters on the Deep Plowed treatment. These trends are graphically exhibited in Figures 4.3.1 and 4.3.2. Metal loadings for vegetation grown on the coversoil were generally similar to concentration loadings for plants grown in the treated plots at the Rocker, Fairmont and Opportunity sites. At Ramsay Flats, however, metal loadings were elevated for plants in the treated plots compared to those in the coversoil.

To designate the plant species that performed well on each waste type, Importance Values were generated. This number provides an objective, data based measurement of the *importance* of each seeded species at each site (Table 4.3.18). These importance species were easy to establish and provided better cover and production than the other seeded species. Longterm survival of these species cannot be determined at this time. Metal levels in and on these species at Sites 7, 27, and 33 were generally

Table 4.3.18. Species exhibiting highest Importance Values.

Site	Species	Overall Rating ¹
7	Mammoth wildrye	109.2
	Altai wildrye	103.2
	Thickspike wheatgrass	36.4
21	Tall wheatgrass	184.9
	Intermediate wheatgrass	125.3
	Russian wildrye	59.5
27	Desert wheatgrass	144.9
	Crested wheatgrass	101.3
	Russian wildrye	72.8
33	Altai wildrye	103.7
	Basin wildrye	51.8

¹ Grand mean of Importance Values.

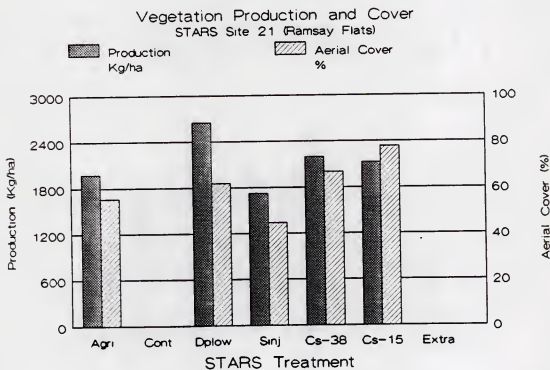
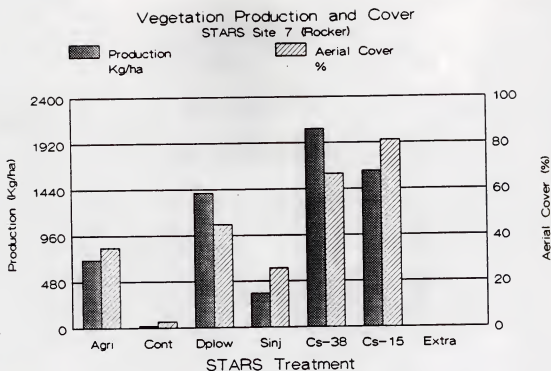


Figure 4.3.1. Vegetative response (production and cover) by reclamation treatment (Agri=Agricultural; Cont=Control; Dplow=Deep Plow; Sinj=Slurry Injection; CS-38 and CS15=Coversoil at 38 cm and 15 cm depths; and Extra=Extra phosphorus fertilizer) for Sites 7 and 21.

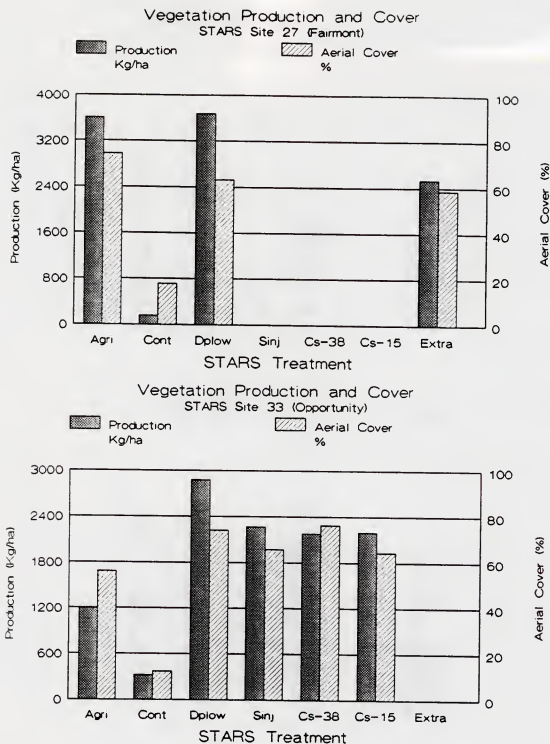


Figure 4.3.2. Vegetative response (production and cover) by reclamation treatment (Agri=Agricultural; Cont=Control; Dplow=Deep Plow; Sinj=Slurry Injection; CS-38 and CS15=Coversoil at 38 cm and 15 cm depths; and Extra=Extra phosphorus fertilizer) for Sites 27 and 33.

within ranges recommended for livestock consumption. Cadmium was the notable exception. Concentrations of this element often exceeded recommended tolerances for livestock consumption. Metal and As concentrations were very high for dominant species at Site 21, and it is inferred that these high metal levels were due to dust and rainfall splash. Contaminants were on the plants rather than in the vegetation.

4.4 IMPLEMENTABILITY

The success of any large scale reclamation/treatment project depends not only on the effectiveness of the treatment method selected, but the ability to efficiently implement those treatment methods in the field as well. In other words, the most effective treatment methods under laboratory conditions may not necessarily be the best option for larger field applications if parameters are involved that render the treatment method too costly or impractical.

To be effective, the treatment method must be versatile enough to meet a variety of field conditions while maintaining the primary goal, namely, thorough mixing of chemical amendments to a depth sufficient to support vegetative growth and reproduction. Under actual conditions, tailings depths, gradation, and toxicity can change within small areas. Other variables which must be considered include depth to groundwater, surface (soil) conditions, and accessibility. Each of the four treatment alternatives (ag-till, deep plow, injection, and cover soil) are discussed further in subsections 4.4.1 through 4.4.4 below.

A common denominator to all the treatment options is the use of chemical amendments. The logistics of supplying and handling large quantities of these materials (generally considered as hazardous) can pose significant problems unless properly planned for. Problems associated with the use of dry lime (or other dry chemical amendments) include supply, transportation, dust, accessibility, stockpile management, health risk, and public access to hazardous materials. Dry lime should not be applied during windy conditions.

Liquid chemical amendments present many of the same problems associated with dry amendments, the exception being limited dust. However, they present other unique problems as well. Liquid amendments are bulky and heavy making handling a problem, particularly in wetter or unstable areas. Storage requirements for many liquid chemical amendments are costly and difficult to obtain. In addition, liquid lime may cause serious skin or eye problems if mishandled. The implementation of chemical amendments will be further discussed in following subsections.

4.4.1 Agricultural Tillage (Ag-Till)

For tailings with maximum depths of approximately 10 to 12 inches, agricultural type tillage methods (ag-till) are considered to be the most cost effective (section 4.5), provide the most thorough treatment, and have the highest degree of "implementability" of any of the treatment options utilized during this study. However, for tailings depths exceeding 10 to 12 inches, ag-till is not effective due to the limited depth of possible amendment incorporation.

Ag-till methods can be readily adapted to a variety of conditions, provided tailings depths remain within the acceptable range. A wide range of relatively low cost equipment is available throughout the U.S. and much of abroad. With proper equipment selection, almost any surface or subsurface condition can be effectively dealt with, hence virtually no specialized equipment is required. Ag-till equipment is relatively maneuverable, allowing for use in constricted areas. In addition, ag-till is the fastest of the treatment methods, and can provide surface conditions conducive to seeding. As with all treatment methods, a system to apply the chemical amendment(s) is required. This equipment is generally available and usually consists of dry fertilizer placement equipment such as spinner, air-flow or conveyor type systems capable of accurate calibration. Fluid lime applicators capable of continuous mixing may also be an option.

4.4.2 Deep Plow

For deeper tailings (up to four feet in depth), deep plowing is the most effective of the selected treatment methods, and the least expensive (of the deeper treatment options). Although not quite as readily available as standard agricultural tillage equipment, several brands and types of deep plows are commercially available. Considerable horsepower is required to operate these implements. Specialized equipment is not required. The deep plow tested was a single bottom unit called the Dika plow. This unit had sufficient tillage depth (four feet), and provided the most thorough mixing at higher amendment rates. This unit was most effective when combined with an ag till "finish".

The deep plow treatment method has several drawbacks. The deep plows are generally bulky, slow, and less maneuverable (than ag till). They require considerably higher power requirements (in the range of 300 horsepower), and tend to be higher maintenance. Deep plowing leaves a roughened finish surface requiring surface prep prior to seeding, however, when combined with the recommended final pass with ag till equipment, this no longer presents a problem. Finally, when working in poorly drained areas, the operation and degree of mixing become marginal. Deep plows are able to provide an added benefit by breaking up compacted layers which might interfere with plant root growth.

4.4.3 Injection Treatment

Lime slurry injection treatment is capable of providing the greatest depth of treatment (up to 40 feet), however, it is considered the least effective, least implementable, and most expensive of the deeper treatment methods utilized during the study. Generally, injection treatment is effective only under optimal conditions. Although the equipment used to effectively complete this method of treatment is relatively specialized, it is commercially available. Equipment can be adapted to be highly maneuverable provided the method for supplying lime to injection equipment is equally as maneuverable. Lime slurry is difficult to handle, therefore equipment for mixing, transporting and handling the lime slurry is also

necessary. Injection treatment requires a surface treatment with ag-till equipment, not because of surface conditions, but rather for the untreated surface. Logistics can be a problem, as well.

As stated above, optimal conditions are necessary for effective treatment. Soils (tailings) that are rocky, tight, or inconsistent reduces or eliminates the success of the treatment. Excavation at the STARS plots to evaluate amendment mixing revealed the injection method as the least effective at mixing amendment throughout the treated soil profile (section 4.2.4). Application rates for the amendments are difficult to control, and many amendments are not compatible with injection systems. When conditions are such that amendment rates are higher than stoichiometrically necessary, the costs are even higher, and solidification of the soil may occur. This method does allow for selective depth placement of amendment which may be applicable at select locations, and treatment of wetter tailings is possible.

4.4.4 Cover Soil Treatment

This treatment option is more of an "isolation-neutralization" method than a standard "treatment-incorporation" method. It involved an ag till treatment followed by placement of varying depths of cover soil. As with the standard ag-till alternative, a wide range of equipment is readily available that is capable of completing the necessary work. By selecting the proper equipment, problems associated with maneuverability, access, and wet conditions can be virtually eliminated. This will tend to keep equipment costs down, however, chemical amendment and cover soil import costs are still necessary. Depending on the availability, haul distance, and purchase price, the actual cost of the cover soil treatment method may be the most costly.

For a large scale project, the volume of coversoil necessary to complete the work would be substantial. Locating borrow sources within a reasonable haul distance and with sufficient volumes will most likely be difficult to accomplish. However, this method may be useful for treatment of small areas that are not amenable to treatment by other methods due to high groundwater levels or poor soil characteristics.

4.4.5 Implementability Summary

The ag-till, deep plow and cover soil treatments are the most easily implemented of the treatments tested by virtue of the lack of any specialized equipment necessary for these methods. The cover soil treatment may be burdened by logistics of acquiring a sufficient supply of coversoil and has associated costs significantly greater than the ag-till or deep plow treatments (section 4.5). The injection treatment does require specific equipment which may limit its use. The ag-till and deep plow methods would appear to be applicable for use over large areas, with the cover soil method to a lesser extent.

4.5 COSTS

Construction costs, in addition to treatment effectiveness and implementability, are a critical component to any treatment option, regardless of the size, nature and complexity of the project. Alternatives for the treatment of mine tailings must be cost effective in order to remain as a viable reclamation/stabilization technique.

Two of the STARS treatment methods, namely the ag-till and deep plow alternatives, were implemented as part of the Clark Fork River Demonstration project completed in 1990 near Warm Springs, MT. This project centered around demonstrating the effectiveness of these , and other, *in-situ* reclamation techniques to neutralize and stabilize large scale toxic mine tailings within the floodplain of the Clark Fork River system. From this project, actual construction costs (1990 prices) associated with these techniques were determined. These actual costs will be compared with the estimated costs found in Section 7.0 of the Final Summary Report for Phase II of STARS (Reclamation Research Unit et al, 1989a). The remaining two methods, cover soil and lime slurry injection treatments, have not been demonstrated in actual large scale projects. Estimated costs from the STARS Phase II summary report, referenced above, will be used for this evaluation. Some decrease in the unit costs of these latter treatments would also be expected if implemented on a large scale.

4.5.1 Ag-Till

Actual construction costs for ag-till treatment method (1990 prices) are summarized in Table 4.5.1 below. Estimated construction costs from the STARS Phase II summary report follow in Table 4.5.2. Similar work items were included in both tables. Area grading is the same manner of work as mellowing-tillage. The unit cost/acre does not include costs for engineering/design, mobilization, clearing and grubbing, fencing or support equipment. It should be noted that neither phosphogypsum or ferric sulphate were used on the Clark Fork demonstration site, however, these items do not account for the approximate \$1600/acre difference from the STARS Phase II summary report estimate. It is believed that the savings is due to lower unit costs associated the larger scale project.

4.5.2 Deep Plow

Actual construction costs for the deep plow treatment (1990 prices) are summarized in Table 4.5.3. Estimated construction costs from the STARS Phase II report follow in Table 4.5.4. Similar work items were included in both tables, with area grading the same manner of work as mellowing-tillage. The unit cost/acre does not include costs for engineering/design, mobilization, clearing and grubbing, fencing or support equipment. The large scale deep plow treatment did not include phosphogypsum or ferric sulfate. These costs represent only 12 percent of the estimated STARS Phase II summary report costs and, similar to the ag-till, most of the cost reduction is believed to be due to lower unit costs associated with the larger scale project.

Table 4.5.1. Actual construction costs for a large scale ag-till treatment project.

AG-TILL TREATMENT METHOD ¹			
Cost Item	Actual Quantity	Actual Unit Cost	Total Cost (\$/Acre)
Lime (CaO)	5.6 Ton/Acre	\$ 86.50/Ton ²	484.40
Lime (CaCO ₃)	41.7 Ton/Acre	\$ 32.50/Ton ²	1354.40
Incorporation Ag-Till ³	Lump Sum	\$ 196/Acre	196.00
Water	5.1 Kgal/Acre	\$ 12/Kgal	61.20
Area Grading	Lump Sum	\$ 150/Acre	150.00
Fertilize and Seed	Lump Sum	\$ 325/Acre	325.00
Vegetative Mulch	Lump Sum	\$ 325/Acre	325.00
TOTAL			2896.00

¹ Assumes a treatment depth of 0 - 8".

² Includes all materials, handling, transportation and application costs.

³ Includes two perpendicular passes.

Table 4.5.2

Cost estimate for implementation of the agricultural incorporation treatment, from STARS Phase II Final Report, Section 7.0.

AG-TILL TREATMENT METHOD ¹			
Cost Item	Typical Amount	Estimated Unit Cost	Total Cost (\$/Acre)
Lime (CaO)	40 tons/acre	\$40/ton	1600
Lime (CaCO ₃)	40 tons/acre	\$70/ton	700
Phosphogypsum	5 tons/acre	\$30/ton	150
Ferric Sulphate	2 tons/acre	\$200/ton	150
Water	50,000 gallons	\$20 /1000 gal	1000
Mellowing-Tillage	1.0 hr/acre	\$60 /hr	65
Lime Spreader	3 hours/acre	\$80 /hr	240
Lime Nurse Truck	1.5 hours/acre	\$45 /hr	65
Amendment Incorporation	1.0 hr/acre	\$60 /hr	60
Seedbed Preparation	1.0 hr/acre	\$60 /hr	60
Seeding	0.5 hr/acre	\$80 /hr	40
Seed and Fertilizer	lump sum	\$125/acre	125
TOTAL			4,500.00

¹ - Assumes that the 0-8 inch layer is amended.

Table 4.5.3. Actual construction costs for a large scale deep plow treatment.

DEEP PLOW TREATMENT METHOD¹			
Cost Item	Actual Quantity	Actual Unit Cost	Total Cost (\$/Acre)
Lime (CaO)	31.3 Ton/Acre	\$ 86.50/Ton ²	2,707.50
Lime (CaCO ₃)	233.3 Ton/Acre	\$ 32.50/Ton ²	7,582.30
Incorporation Deep Plow ³	Lump Sum	\$1,015/Acre	1015.00
Incorporation Ag-Till ³	Lump Sum	\$ 196/Acre	196.00
Water	5.1 Kgal/Acre	\$ 12/Kgal	61.20
Area Grading	Lump Sum	\$ 150/Acre	150.00
Fertilize and Seed	Lump Sum	\$ 325/Acre	325.00
Vegetative Mulch	Lump Sum	\$ 325/Acre	325.00
TOTAL			12,362.00

¹ Assumes a treatment depth of 0 - 48".

² Includes all material, handling, transportation and application costs.

³ Includes two perpendicular passes.

Table 4.5.4 Cost estimate for implementation of the deep mechanical tillage method, from STARS Phase II Final Report, Section 7.0.

DEEP PLOW TREATMENT METHOD¹			
Cost Item	Actual Quantity	Actual Unit Cost	Total Cost (\$/Acre)
Lime (CaO)	240 tons/acre	\$40/ton	9600
Lime (CaCO ₃)	60 tons/acre	\$70/ton	4200
Phosphogypsum	30 tons/acre	\$30/ton	900
Ferric Sulphate	12 tons/acre	\$200/ton	2400
Initial Tillage	2 hrs/acre	\$60 /hr	120
Deep Tillage ²	1.5 hrs/acre	\$150 /hr	225
Water	80,000 gallons	\$20 /1000 gal	1600
Mellowing-tillage	2.0 hr/acre	\$60 /hr	120
Lime Spreader	18 hours/acre	\$80 /hr	1440
Lime Nurse Truck	9 hours/acre	\$45 /hr	405
Seedbed Preparation	0.5 hr/acre	\$80 /hr	60
Seeding	0.5 hr/acre	\$80 /hr	40
Fertilize and Seed	lump sum	\$125/acre	<u>125</u>
TOTAL			21,235.00

¹ Assumes a treatment depth of 0 - 48".

² Includes D-8, deep plow rental and mobilization costs.

4.5.3 Injection Treatment

The injection treatment was employed only on the STARS plots and therefore no large scale project costs are available. The STARS Phase II cost estimates indicate the method is approximately 50 percent more expensive than the deep plow treatment (\$31,517/acre versus \$21,236/acre). It is likely that a savings of approximately 40 percent, as found for the ag-till and deep plow treatments, may also be possible for a large scale injection treatment project.

Table 4.5.5 Cost estimate for implementation of the Lime Slurry Pressure Injection (LSPI) deep incorporation method, from STARS Phase II Final Report, Section 7.0.

LIME SLURRY PRESSURE INJECTION TREATMENT METHOD¹			
Cost Item	Actual Quantity	Actual Unit Cost	Total Cost (\$/Acre)
Lime (CaO)	240 tons/acre	\$40/ton	9600
Lime (CaOH ₂)	60 tons/acre	\$70/ton	4200
Phosphogypsum	5 tons/acre	\$30/ton	400
Ferric Sulphate	2 tons/acre	\$200/ton	400
LSPI ²	Lump Sum	\$16062/acre	16062
Post-LSPI Tillage	1 hr/acre	\$60/hr	60
Topdress-Lime Spreader	4.5 hours/acre	\$80/acre	360
Water-Mellowing	20000 gallons	\$20/1000 gal	400
Mellowing-tillage	1.0 hr/acre	\$60 /hr	60
Seedbed Preparation	1.0 hr/acre	\$60 /hr	60
Seeding	1.0 hr/acre	\$60 /hr	40
Fertilize and Seed	lump sum	\$125/acre	125
TOTAL			31,517.00

¹ Assumes that all amendments are added to the 0 to 48 inch layer.

² Includes rental of LSPI injector and batch tanks including labor and mobilization, injection to be placed on 2.5 foot centers.

4.5.4 Cover Soil Treatment

The cover soil treatment was not used in any large scale projects and the costs presented are those given in the Stars Phase II summary report (Reclamation Research Unit et al, 1989a). Again, it is likely that unit costs associated with a large scale implementation of this treatment will be somewhat less than those presented (Table 4.5.6).

Table 4.5.6 Cost estimate for implementation of the coversoil treatment method, from STARS Phase II Final Report, Section 7.0.

COVER SOIL TREATMENT METHOD¹			
Cost Item	Actual Quantity	Actual Unit Cost	Total Cost (\$/Acre)
Lime (CaO)	40 tons/acre	\$40/ton	1600
Lime (CaCO ₃)	10 tons/acre	\$70/ton	700
Phosphogypsum	5 tons/acre	\$30/ton	150
Ferric Sulphate	2 tons/acre	\$200/ton	400
Water	50,000 gallons	\$20 /1000 gal	1000
Mellowing-tillage	1.0 hr/acre	\$80 /hr	60
Lime Spreader	3 hours/acre	\$80 /hr	240
Lime Nurse Truck	1.5 hours/acre	\$45 /hr	65
Amendment Incorporation	1.0 hr/acre	\$60 /hr	60
Coversoil ²	1610 cy	\$5.50 /cy	8,855
Coversoil Spreading (D8)	1.0 hr/acre	\$125 /hr	125
Seedbed Preparation	1.0 hr/acre	\$60 /hr	60
Seeding	0.5 hr/acre	\$80 /hr	40
Fertilize and Seed	lump sum	\$125/acre	<u>125</u>
TOTAL			13,480

¹ Assumes that the 0-8 inch layer is amended and is then buried by coversoil.

² Assumes that 12 inches of coversoil is used. The cost estimate is based on bank yardage (hailed yardage = 1.5 x bank yards).

4.5.5 Cost Summary

The costs for the four treatments presented in the STARS Phase II summary document and the two large scale treatments are summarized in Figure 4.5.1. The ag-till treatment is the least costly to implement while the injection method is the most expensive. If sources of coversoil are readily available, this treatment may be preferable to the deep plow treatment based on costs.

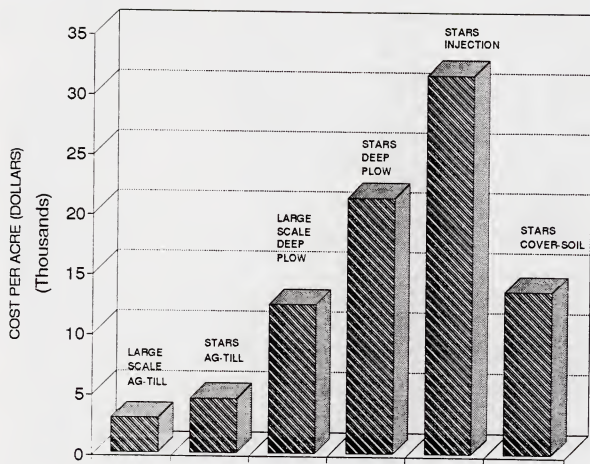


Figure 4.5.1. Cost summary of STARS treatment alternatives.

5.0 DATA INTERPRETATION AND MODELING

Results of Phase III monitoring efforts were used to describe the short-term effectiveness; reduction in mobility and toxicity; cost; and implementability of STARS treatments. The duration of monitoring was insufficient to assess the potential long-term effectiveness of STARS treatments. Monitoring results can be used to calibrate physical models of key transport processes, however. Using calibrated models, the long-term environmental performance can be evaluated.

Three key physical and chemical processes are thought to control the mobility of heavy metals in streambank tailings; surface water runoff causing transport of sediment and dissolution of metal-enriched salts; flux of contaminated soil water through the vadose zone into groundwater; and chemical reactions involving metals in contaminated soil horizons and in underlying natural soils and sediments involving speciation, dissolution/precipitation and adsorption. The results of modeling each of these processes are described in this section.

Surface water runoff and erosion was modeled using GLEAMS Version 2.0 (Knisel et al. 1992) developed by the U.S. Department of Agriculture Agricultural Research Service (ARS). Convective transport of soil water through the unsaturated zone in which the majority of tailings reside and into shallow groundwater was modeled using UNSAT2, a code developed by Neuman for the Department of Energy. Finally, various geochemical equilibrium processes which may occur in amended and unamended tailings, in buried soil layers, and in surface runoff were modeled using MINTEQA2, an equilibrium speciation model developed for EPA (Allison et al. 1991).

5.1 SURFACE RUNOFF AND EROSION

5.1.1 Historical Discussion of Rainfall/Runoff and Erosion Models

A number of candidate models are available for simulating rainfall-runoff response and erosion. Many of the available models have a common origin with GLEAMS and use similar mathematical routines for predicting and routing runoff and for estimating erosion and sediment transport. The CREAMS model (Knisel 1980), the precursor of GLEAMS was first released in 1980 after extensive model validation at numerous ARS facilities throughout the continental U.S. A discussion of numerous hydrologic/erosion models (Beasley 1989) is contained in the proceedings from the CREAMS/GLEAMS Symposium (Beasley, Knisel and Rice, 1989).

Sedimot II (Wilson et al. 1982), developed at the University of Kentucky is a watershed-scale event-based model primarily used for evaluating management and design effects on erosion and sediment transport. Runoff is estimated using the same SCS curve number approach implemented in GLEAMS. Soil loss can be predicted using either of two methods, one of which is the modified Universal Soil Loss Equation (USLE) as in GLEAMS. Overall watershed response is simulated by using a Muskingham routing technique for flow from homogeneous subwatersheds.

The Pesticide Root Zone MODEL (PRZM) developed by EPA (Carzel et al. 1984) is a continuous field-scale model. Both PRZM and GLEAMS were developed to predict climatically-driven runoff, erosion, nutrient and pesticide movement. While the models differ in their approach to simulating chemical fate and transport, their hydrology and erosion subroutines are very similar.

The ANSWERS model (Beasley, Huggins and Monke 1980) developed at Purdue University is a basin-scale, event-based model. Watersheds are depicted by areal grids, each of which contain specific topographic, soil, and land use inputs. ANSWERS is data-intensive and requires input of breakpoint (eg. hourly) rainfall data. These requirements limit the applicability of ANSWERS for problems where data input are available or can be collected for the basin of interest.

The EPIC model (Williams 1989) is a specialized modeling tool developed from the CREAMS / GLEAMS code that assesses the long-term (100 to 1000 year) impact of soil erosion on soil productivity. The model was developed primarily as a planning tool to investigate the long-term ramifications of various cropping systems and agricultural management techniques. A weather generation component and a wind erosion subroutine were added to the GLEAMS hydrology and erosion code to develop EPIC.

Based upon an evaluation of the available modeling packages, the GLEAMS model appeared to provide the best combination of features including:

- Extensive model validation;
- Field-scale rather than basin-scale domain;
- Physically-based parameter selection thus simplifying the model calibration process;
- Widely accepted simulation approach including the curve number approach for runoff simulation, a modified USLE for soil loss/detachment, and use of the Yalin model for sediment transport;
- Continuous rather than event-based simulation.

5.1.2 GLEAMS Model Theory and Structure

The GLEAMS model consists of several component submodels for calculation of root zone hydrology, erosion, nutrient flux, and pesticide flux. Separate parameter input files are developed to run each component. Pass files are created by each component run for use in subsequent batch routines. Only the hydrology and erosion components of the GLEAMS model were used for this simulation.

The first component, hydrology, uses daily rainfall data, monthly temperature and solar radiation data, and various soil parameters for computation of the daily water balance. The amount and timing of runoff as well as other components of the water balance are computed, and pertinent information on storm size and runoff are passed to the erosion model component.

The technique used for estimation of runoff from daily rainfall data is the SCS curve number approach which has been widely adopted throughout North America. The curve number approach (Mockus 1985) relates the depth of runoff for a given depth of rainfall to the antecedent soil water content and to the "curve number". Curve numbers vary from 0 to 100, and are related to the infiltration capacity of the soil. Detailed guides have been developed for estimating curve numbers (Mockus, 1985)

Runoff is numerically related to rainfall by [1]. During a rainfall event, no runoff is presumed to occur until the rainfall depth exceeds 0.2 of the remaining soil water storage capacity. Soil storage [2] is related to the relative degree of soil saturation (which is influenced by historical rainfall and evaporation) and by the maximum storage [3] which is inversely proportional to the curve number for condition I. Condition I is the curve number for dry soil moisture conditions. While curve numbers for condition II (average soil moisture) are tabulated in USDA references, CN(I) values corresponding to CN(II) values can be calculated using [4].

$$Q = \left[\frac{(P-0.2s)^2}{(P+0.8s)} \right] \quad [1]$$

where Q = Runoff depth (inches)

P = Rainfall depth (inches)

s = soil storage coefficient (inches)

$$s = s_{\max} \left[\frac{(UL-SM)}{UL} \right] \quad [2]$$

where s_{\max} = see equation [3]

UL = Upper limit of soil water storage (inches)

SM = Current stored soil moisture (inches)

$$S_{\max} = \left[\frac{1000}{CN(I)} - 10 \right] \quad [3]$$

$$CN(I) = -16.91 + 1.348[CN(I)] - 0.01379[CN(I)^2] + 0.0001177[CN(I)^3] \quad [4]$$

Runoff volumes for increasing rainfall amounts are shown in Figure 5.1.1. High curve numbers are typical of impermeable soils and disturbed areas while low curve numbers are typical of well-vegetated permeable natural soils. Implicit in the SCS curve number method is that runoff initiation occurs after 0.2 times the storage. In many urban areas where pavement and bare soils are common, runoff occurs during much smaller rainfall events. In addition, infiltration is assumed to decrease to zero after a rainfall depth equivalent to s has occurred. This assumption too may be erroneous especially for permeable soils. Nonetheless, the SCS curve number method has been widely validated using research watersheds from throughout the U.S.

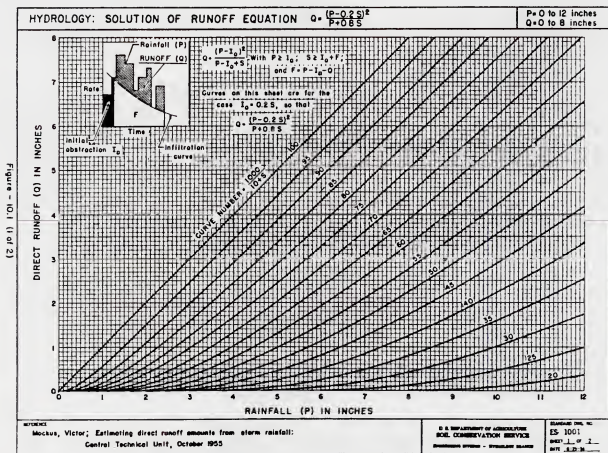


Figure 5.1.1. Rainfall-runoff relationships for soils with varying curve numbers (from SCS National Engineering Handbook, Section 4).

While the SCS curve number approach relies on a generalized assessment of the antecedent moisture condition, the GLEAMS model maintains a daily water balance so that s can be computed. In addition, the water balance module can keep track of cumulative runoff, evaporation, transpiration, percolation, and changes in soil water content [5].

Daily evaporation is the sum of soil and plant evaporation. The maximum daily potential evaporation (PET) [6] is based on the mean daily temperature and solar radiation, each of which are input as mean monthly values in the GLEAMS model. The PET calculated by the Ritchie equation is generally higher than that calculated by the solar thermal unit equation selected for PET calculation in the STARS work plan (Schafer and Associates and MSU, 1990).

$$P = RO - ET - PERC \pm SOIL\ WATER \quad [5]$$

$$E_o = \left[\frac{1.28 \delta \eta_o}{\delta \gamma} \right]$$

E_o = potential daily evaporation (inches)

$$\delta = \left[\frac{5304}{T^2} \right] e^{21.255 - \frac{5340}{T}}, T \text{ in degrees Kelvin} \quad [6]$$

$$\eta_o = \frac{(1-\lambda)(R)}{58.3}$$

λ = albedo

R = solar radiation (langley/day)

γ = psychrometric constant

Soil evaporation [7] is calculated as a negative exponential function of the leaf area index (LAI). Hence, as the plant canopy coverage increases, soil evaporation decreases in importance. Prior research on soil evaporation from bare soil surfaces (Hillel, 1980) has shown that it occurs as a two-stage process. When the soil surface has a high water content, soil evaporation continues at a rate limited by the potential evaporation (E_o). When the upper few millimeters of the soil surface dries, however, the soil evaporation rate decreases due to the slow rate of vapor phase transmission of water. The two-stage soil evaporation process utilized by the GLEAMS model is given in [8]. Values for α_s for different soil textures are given in the GLEAMS manual. When the calculated soil evaporation exceeds ϵ_s , the slower stage of soil evaporation is triggered.

$$E_s = E_o e^{-0.4 LAI} \quad [7]$$

Plant evapotranspiration is assumed to be a linear-weighted fraction of the PET for LAI values up to 3.0. When soil water is limited (less than 25 percent of available water), the plant evaporation decreases linearly until the soil is dry. Values of LAI can either be provided by the GLEAMS pre-processor for over 75 agronomic crop and forage species or can be input as a function of growing season length and julian date. LAI values of zero default to simple bare soil evaporation. Soil water utilized by plants is extracted preferentially from upper soil layers. As the growing season progresses, water extraction can occur at progressively greater depth. This algorithm simulates root extension of annual crops and root activity of perennial crops.

$$\begin{aligned}
 E_s &= E_o e^{-0.4LAI} \text{ where } \epsilon_s < E_o \\
 E_s &= \alpha_s \left[t^{0.5} - \left(t - \frac{1}{2} \right)^{0.5} \right] \text{ where } \epsilon_s > E_o \\
 \epsilon_s &= 9(\alpha_s - 3)^{0.42} \\
 \epsilon_s &= \text{maximum daily soil evaporation} \\
 \alpha_s &= \text{evaporation coefficient}
 \end{aligned}
 \tag{8}$$

$$\begin{aligned}
 E_p &= \left[\frac{(E_o LAI)}{3} \right] \text{ for } LAI < 3 \\
 E_p &= E_o - E_s \text{ for } LAI > 3
 \end{aligned}
 \tag{9}$$

The erosion component of the GLEAMS model utilizes the Yalin equation to compute the sediment transport capacity of runoff. Estimated soil loss can either be "detachment-limited" or "transport-limited". Detachment can occur either due to rain-drop splash or due to the energy of overland flow.

Overland flow from hillslopes can be routed in a number of ways depicted in Figure 5.1.2. The simplest simulation is for overland flow to be routed directly to a channel at the edge of the domain being modeled. In addition, overland flow can be concentrated into a channel within the field boundary. Finally, the hillslope profile can be segmented into various shapes (eg. uniform, convex, concave, or complex). Overland flow and sediment is routed through each segment. Deposition or flow-induced detachment can occur within any hillslope of channel element within the model domain.

The GLEAMS model also predicts the particle size distribution of eroded sediment. When deposition occurs within the field area or channel or when transport capacity is exceeded the transported sediment may consist on average of finer particles than the average grain size in surface soils.

5.1.3 GLEAMS Model Validation

The hydrology and erosion components of the GLEAMS model have been validated at experimental research watersheds in Montana, Texas, Oklahoma, Ohio, Georgia, Nebraska, West Virginia, Mississippi, Iowa, Arizona, and New Mexico. In general, the long-term trend in runoff and erosion rates were accurately predicted by GLEAMS although model performance on individual storms was less reliable.

5.1.4 GLEAMS Model Calibration

Parameter Estimation: The GLEAMS model requires calibration before it can be used to simulate runoff and erosion from unknown areas. Due to the complexity of the GLEAMS model, only the control and the most successful amendment/revegetation treatment at each location were simulated using GLEAMS.

Precipitation input for GLEAMS runs was from the Anaconda, MT NOAA weather station. A comparison of rain gauge measurements from each of the sites indicated that the Anaconda site correlated well with other stations in the basin and tended to have somewhat higher cumulative precipitation. Three full years of input data (1989 through 1991) were used in the GLEAMS simulations. Mean monthly average temperature and solar radiation values were calculated for the Ramsay Flats climate station for the entire period of record. GLEAMS results are not highly sensitive to small variations in daily temperature and solar radiation, hence monthly inputs were used.

Input parameters for the hydrology and erosion simulations are listed in Table 5.1.1, and input parameter files are provided in Appendix E-1. Values for all parameters were either input as default values, or were measured. Input data including batch input file listings are summarized for each GLEAMS run in Appendix E-1. The most sensitive parameter in the GLEAMS model is the runoff curve number. Due to the sensitivity of this parameter, great care was taken in estimating CN values. Rainfall simulation test data were used to calculate CN values (Section 4.1). Rainfall-runoff data from Ramsay Flats, Opportunity, and Rocker were collected during summer, 1992 (Tables 4.1.1 through 4.1.6). The control plots at Ramsay Flats and Opportunity had unusually high curve numbers presumably because of the rainfall-induced compaction on the exposed tailings. A curve number of 95 was used for the control plots at each site. The coarse texture of tailings at Rocker resulted in a control curve number estimated to be 55. Measured curve numbers for Ramsay Flats and Opportunity were 94 and 93 while the Rocker site had a CN of less than 50. Due to the extremely high infiltration rate, no runoff occurred at Rocker during the rainfall simulation tests. The deep-till/revegetated plots at Ramsay Flats and Opportunity had much lower measured CN values than the control (59 and less than 50) due to the effects of tillage and revegetation. The curve numbers selected for the GLEAMS simulation (78 at Ramsay Flats and 65 at Opportunity) were conservatively set higher than the measured CN values due to potential bias in site selection for the rainfall simulation tests.

In general, a relatively thin root zone depth was input to simulate the control plots because observed changes in soil water content due to evaporation were confined to the upper 12 inches of soil. Values for hydraulic conductivity, and soil water-holding capacity were based on measurements taken at each site.

The leaf area index (LAI) values and the crop type are important variables in simulating the on-site water balance. For the control plots, a LAI value of near zero was input so that GLEAMS would default to soil evaporation. For the vegetated plots, numerous preliminary GLEAMS runs were performed to try to simulate the observed changes in soil water content from the site. Very high values of LAI had to be input in order for plant evapotranspiration to approach soil evaporation in magnitude. Despite the fact that actual LAI values on the deep-till plots are near 1 or less, higher LAI values were input to improve model results. The beginning and end of the growing season and relative seasonal LAI values were input to simulate actual conditions.

GLEAMS Calibration Runs: Before proceeding with GLEAMS simulation, it was necessary to try to compare GLEAMS model output to the Silver Bow Creek data set. GLEAMS has been widely validated at various locations throughout the U.S. The purpose of on-site calibration was to identify the accuracy of GLEAMS for contaminated streambank tailings areas, and to identify the accuracy of selected input parameters.

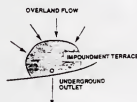
While measured runoff and erosion data are unavailable from the STARS plots, GLEAMS can be calibrated using the measured amount of stored soil water within each of the plots modeled. Calibration was performed for the control and deep-till plots at Ramsay Flats (Figure 5.1.3). The quantity of stored water measured in the upper 150 to 250 cm of the soil profile varied seasonally from a low in late summer to a peak in late spring. The soil profile was recharged gradually between September and April while maximum water content, near field capacity, was usually observed between mid-April and early June. The rate and total quantity of water withdrawn by plants varied markedly between treatments with less than 1 inch of withdrawal common in the control plot to over four inches in the deep plow plot.

In the GLEAMS Ramsay Flats control plot simulation (Figure 5.1.4, Table 5.1.2-Run 1), the water content remained at or near the wilting point from January through March and gradually increased to a peak sometime in May. The total annual variation in water content (0.5 inches) was less in the simulation than the measured amount. For the average annual water balance, 1.51 inches of runoff and no percolation were predicted for 12.8 inches of precipitation. Soil loss averaged 13.7 tons/acre/year.

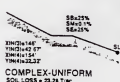
For the GLEAMS Ramsay Flats deep plow simulation (Figure 5.1.5, Table 5.1.2, Run 2), differences between the observed and simulated soil water balance were more pronounced than for the control. Annual variation in water content for the GLEAMS simulation of a well-vegetated plot were on the order of 0.5 inches, much less than the four



Option 1 - Overland flow sequence and slope representation



Option 2 - Overland flow sequence



Option 3 - Overland flow sequence

Figure 5.1.2. Selected options for routing overland flow in the GLEAMS model.

Table 5.1.1. Input parameters used for the GLEAMS model - Silver Bow Creek, Montana.

GLEAMS MODEL DOMAIN	
HYDROLOGIC PARAMETERS	EROSION PARAMETERS
<p><u>Area</u> = 1,000 by 1,000 trapezoidal watershed</p> <p><u>Saturated Hydraulic Conductivity</u> = measured at each site</p> <p><u>Soil Profile Percent Full</u> = Initial water content 65% of field capacity</p> <p><u>Evaporation Coefficient</u> = varies by site</p> <p><u>SCS Curve Number</u> = varies by site, based on rainfall simulation</p> <p><u>Root Zone Depth</u> = varies by site, generally 6 inches for control and 36 inches for vegetated</p> <p><u>Soil Characteristics</u> = porosity, field capacity, wilting point, organic matter percent, clay, and silt content varies by site, based on observed soil morphology</p> <p><u>Monthly Mean Daily Maximum and Minimum Temperature</u> = Based on Ramsay Flats climate station</p> <p><u>Monthly Mean Daily Solar Radiation</u> = Based on Ramsay Flats climate station</p> <p><u>Vegetation Characteristics</u> = varies by site, growing season duration and LAI based on calibration results</p>	<p><u>Slope</u> = Overall 1.5 %, 1,440 feet at 1.5 % then steepening to 30% for last 30 feet</p> <p><u>Watershed shape</u> = length to width ratio 2:1</p> <p><u>Soil Erodibility K Factor</u> = varies by site, based on USLE nomograph</p> <p><u>Cropping Practice P Factor</u> = set to 1.0 for all simulations</p> <p><u>Cover Factor</u> = varies by site, generally 1.0 for control and 0.2 for vegetated</p>

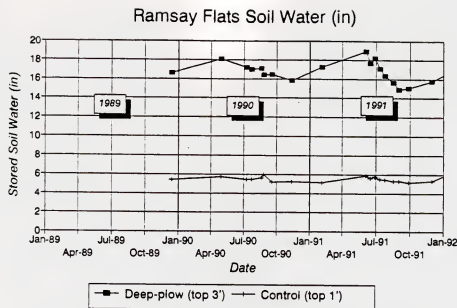


Figure 5.1.3. Stored soil water measured for control and deep-til plots for the Ramsay Flats site.

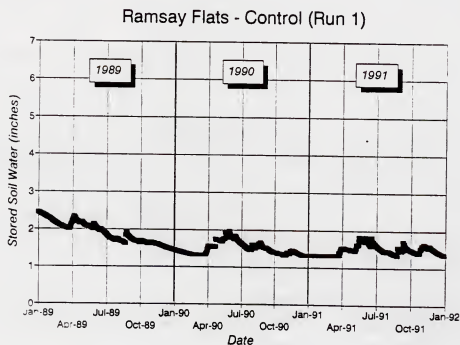


Figure 5.1.4. GLEAMS simulation of stored soil water for the control site at Ramsay Flats.

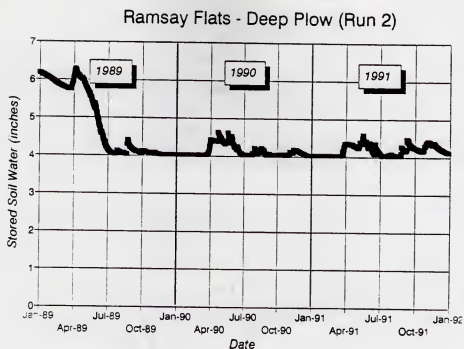


Figure 5.1.5. GLEAMS simulation of stored soil water for the deep-till site at Ramsay Flats.

inch variation actually observed. When the water content was initialized at 75 percent of field capacity, plant transpiration decreased the soil water to the wilting point in the first year and it seldom climbed much above its driest level. Despite performing numerous GLEAMS runs with varying input levels for soil evaporation, plant LAI, and growing season dates, similar results were always obtained. The average annual water balance summary indicated that the deep plow plot would have less runoff (0.77 inches) and less erosion (1.6 tons/acre/year) than the control plot. As for the control, no percolation was predicted for the deep plow plot.

Based on these comparisons, it is evident that the GLEAMS model overpredicts the rate of soil evaporation, underpredicts average soil water content, hence probably underpredicts net percolation. Due to the extensive validation of GLEAMS for other watersheds and its primary use in predicting watershed yield and erosion, it was concluded that predicted rainfall and erosion results were probably more accurate than were the specific terms within the water balance.

The primary cause for overprediction of soil evaporation and underprediction of soil water content (especially between September and June) was the equation used for potential ET [6]. The solar thermal unit equation [10] proposed for use in the Silver Bow Creek investigation was compared to the Ritchie equation for the entire period of record available from Ramsay Flats (Figure 5.1.6). The Ritchie equation predicts much higher daily potential ET values than the solar thermal unit equation during the coldest part of the year, while lower PET is predicted during the summer. Substitution of the solar thermal unit equation into GLEAMS would be expected to greatly improve the models ability to accurately track the soil water balance. Other enhancements to the GLEAMS model that may improve performance may include the method by which water is routed through the root zone. Research is on-going to link a numerical vadose zone model with GLEAMS to satisfy this objective (Bosch 1989).

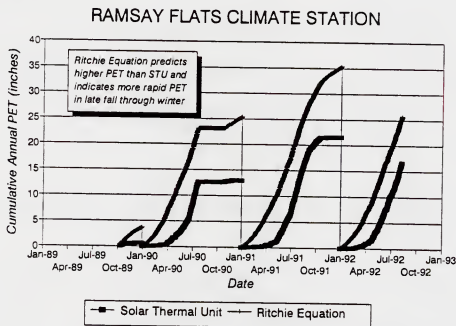


Figure 5.1.6. Comparison of predicted potential evapotranspiration by the Ritchie method and the solar thermal unit equation.

5.1.5 Reliability of GLEAMS Runoff and Erosion Prediction

GLEAMS model results were not rigorously calibrated to measured surface runoff of erosion rates. As a result, the accuracy of the model results are uncertain. Although the absolute quantity of runoff and erosion predicted by the GLEAMS model may not be accurate, relative differences in runoff and erosion between treated and untreated conditions and between sites are probably realistic.

5.1.6 Surface Runoff / Soil Loss Simulation

Ramsay Flats: Runoff and erosion was simulated for the control and deep plow plots at Ramsay Flats using the calibrated GLEAMS model (Table 5.2, Runs 3 through 5, Appendix E-1). Significant differences in the amount of runoff and erosion were predicted between the control and deep plow treatments. For the three year simulation, 1.46 inches of runoff occurred from the control while only 0.67 inches was predicted from the deep plow plot. No percolation below the root zone was predicted for either treatment, hence evaporation and transpiration accounted for the remaining average annual rainfall of 12.8 inches.

The timing of runoff also differed between the control and deep plow treatments. Peak monthly runoff for both treatments occurred in April or May depending on the year. Measurable amounts of runoff occurred throughout the summer from the control plot while no runoff occurred after the end of May from the deep plow plot. This difference in the timing of runoff is thought to be significant in that most fish kills on the Clark Fork have been observed in July and August after convective thunderstorms. Runoff during midsummer may contain higher concentrations of dissolved metals due to the formation of metal-enriched salt crusts in the soil surface during warm weather. In addition, the higher spring instream flow means that runoff from streambank areas is more diluted when it mixes with the channel in spring than in summer.

Significant differences in erosion rates were also noted between the control and deep plow treatments. For the control site, an average annual soil loss of 13.7 tons/acre was predicted while only 1.2 tons/acre was predicted for the deep plow site. The ten-fold reduction in erosion was due to the reduction in runoff as well as the protection provided by the vegetative cover established on the deep plow plots.

Table 5.1.2. Summary of water balance results and soil loss from the USDA CREAMS runoff model for Ramsay Flats (1989 to 1991).

WATER BALANCE TERM	RUN 1 CONTROL	RUN 2 DEEP	RUN 3 CONTROL	RUN 4 DEEP	RUN 5 DEEP
WATER BALANCE SUMMARY					
Precipitation (in/yr)	12.8	12.8	12.8	12.8	12.8
Runoff (in/yr)	1.51	0.77	1.46	0.64	0.67
Evapotranspiration	12.2	13.8	11.8	14.2	14.1
Percolation (in/yr)	0.0	0.0	0.0	0.0	0.0
SOIL LOSS SUMMARY					
Soil Loss (t/acre/yr)	13.7	1.6	13.7	1.1	1.2

Rocker: Similar simulations of the control and deep plow plots at Rocker were also conducted (Table 5.3, Run 6 and 7). The soil material at Rocker was coarser in texture than at Ramsay Flats, hence has a lower water-holding capacity, a much higher infiltration rate, hence was much less erosive than soils at Ramsay Flats. The water balance results for three years of simulated rainfall at Rocker indicated that 1.69 inches of percolation (eg. groundwater recharge) would occur on the control plot, while the higher evaporative use of water by established vegetation in the deep plow plot would prevent percolation. Runoff averaged 0.66 inches and 0.61 inches on the control and deep plow plots respectively. Predicted runoff was less at Rocker than at Ramsay Flats due to the lower runoff curve number of both the control and deep-plow plots at Rocker. Runoff was only predicted in March and April, presumably in response to snowmelt events. Less difference in runoff was noted between the control and deep plow plots because reclamation had less overall effect on the curve number (CN = 40 for both cases) due to the already rapid infiltration on the control plot.

Predicted soil loss at the Rocker site was 0.7 tons/acre for the control and 0.14 tons/acre for the deep plow plot. These low rates of soil loss were due to the coarse texture and rapid infiltration rates of soil at Rocker.

Opportunity: GLEAMS simulation of the Opportunity control site (Table 5.3, Run 8 and 9) yielded an estimated annual 1.69 inches of runoff and 0.42 inches of percolation for 12.8 inches of precipitation. Runoff occurred intermittently from March through September with peak contribution in April. Percolation occurred from April through June when the soil profile was at its annual maximum.

The deep plow plot at Opportunity had substantially less predicted runoff (0.66 inches/year) than the control plot and no percolation. Runoff occurred only during snowmelt in March and April, periods of the year when runoff would be expected to have less impact on fisheries and macroinvertebrates due to the higher in-stream flows. The decrease in runoff was due to the reduction in the curve number observed between bare tailings (curve number = 94) and the deep-plowed and vegetated condition (curve number = 65).

Predicted annual soil loss for the control plot and deep plow treatment was 9.2 and 0.67 tons/acre respectively.

Table 5.1.3. Summary of water balance results and soil loss from the USDA CREAMS runoff model for the Rocker and Opportunity sites Flats (1989 to 1991).

WATER BALANCE TERM	ROCKER CONTROL RUN 6	ROCKER DEEP-TILL RUN 7	OPPORTUNITY CONTROL RUN 8	OPPORTUNITY DEEP-TILL RUN 9
WATER BALANCE SUMMARY				
Precipitation (in/yr)	12.8	12.8	12.8	12.8
Runoff (in/yr)	0.66	0.61	1.69	0.32
Evapotranspiration	12.8	12.9	10.9	12.3
Percolation (in/yr)	1.69	0.0	0.42	0.0
SOIL LOSS SUMMARY				
Soil Loss (t/acre/yr)	0.7	0.14	9.2	0.67

5.2 VADOSE ZONE FLOW

The potential for vertical seepage of soil water through tailings-contaminated soils and into groundwater is of concern at the Silver Bow Creek site. If soil water flows downward through the vadose zone either continuously at a slow rate or intermittently, the potential may exist for metals to reach groundwater. Although groundwater contamination is not widely observed along the streambank areas, the processes which minimize transport of metals to groundwater (eg. lack of seepage or chemical precipitation of metals) were less clear.

Investigation of vadose zone transport was used to characterize the long-term risk of groundwater contamination due to convective transport of soil water. Evaluation of vadose zone seepage separately from geochemical reactions in the vadose zone allowed the importance of each process in minimizing groundwater contamination to be identified. In addition, it was important to compare the rate of seepage through both the untreated and treated tailings. A potential concern regarding STARS treatment is that by increasing soil infiltration rate that shallow groundwater contamination would increase at the expense of surface water protection.

5.2.1 MODEL DESCRIPTION

A number of public domain models were available for simulating variably saturated flow in porous media. UNSAT2 (Davis and Neuman 1983) represents one of the more widely used domain unsaturated flow models. UNSAT2 has been widely validated for use in predicting seepage in agricultural, waste management and containment, and in hydraulic (eg. dams) applications. Although UNSAT2 was used as a 1-D model for this simulation, its 2-D capabilities may be valuable in more complex simulation of groundwater/vadose zone interactions along Silver Bow Creek that may be required in the future. UNSAT2 contains a number of flexible options for simulating the capillary influence of groundwater, evaporation from bare soil surfaces, and withdrawal of water for transpiration by plants.

The numerical basis for unsaturated flow simulation in partially saturated soils is derived from the Buckingham-Darcy flux law [10], (Jury et al. 1991). The first hydraulic head term refers to the soil matric or suction potential. The second term in the hydraulic head expression in [10] is the unity downward gradient for vertical flow. In a soil at equilibrium with a shallow water table h is equal to $-z$ so that the matric potential gradient is -1 and the total head is zero, hence no flux occurs. By measuring the matric potential at a given distance above the water table, the direction of flow in unsaturated soil can be inferred. The soil matric potential always has a negative sign while the suction potential is equal in value to matric potential but is expressed as a positive number. The negative sign in [10] is used so that downward flux will be negative while upward flow is positive. When the matric potential gradient is less than -1 then upward flux occurs. Conversely, when the matric potential gradient is greater than -1 , downward flux will occur.

$$J_w = -K_u \left(\frac{\partial h}{\partial z} + 1 \right)$$

where J_w is the rate of water flux (cm)

[10]

K_u is the unsaturated hydraulic conductivity (cm/s)

h and z are components of total head (cm) such that $(h+z=H)$

The Richards equation which is the fundamental numerical expression governing transient water movement in unsaturated soil is derived by combining [10] with the continuity equation [11]. UNSAT2 like other transient flow models was developed on the basis of the Richards equation with additional source and sink terms to simulate wells, and plant water extraction.

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z} \left[K_u \left(\frac{\partial h}{\partial z} + 1 \right) \right] \quad [11]$$

UNSAT2 version 2.1, written in Fortran-77 is a two-dimensional Galerkin finite element model capable of simulating variably saturated anisotropic flow in heterogeneous materials. Flow regions can be delineated by irregular boundaries. In addition to conventional fixed head, fixed flux, and impermeable boundary conditions, UNSAT2 also provides a number of special boundary conditions controlled by atmospheric variables. Examples include seepage faces, infiltration, and evaporation surfaces. The type of boundary as well as the value of the boundary data can be changed using a simulation restart feature. The relationship between hydraulic conductivity, suction, and water content can be described using the vanGenuchten equation [12] and [13] in version 2.1. Numerical instability can be minimized through use of variable time steps. Source and sink terms within the domain include partially and fully penetrating wells in addition to plant roots. Growth of the plant root zone can be simulated through use of root effectiveness terms that vary with depth and time. Although a storativity term is included in UNSAT2, it is traditionally set to zero as its influence is negligible within the vadose zone.

5.2.2 UNSAT2 CALIBRATION AND DOMAIN

In order to simulate unsaturated flow at the STARS field sites, the hydraulic properties of typical samples were characterized using a variety of field and laboratory tests. Simulation was only performed at the Opportunity, Rocker and Ramsay Flats sites in that these sites were most representative of streambank tailings conditions throughout the operable unit. The Ramsay site represented massive deposits of fine-textured tailings found in the Ramsay area and in other low-gradient stream reaches. Rocker represents tailings deposits in high-gradient reaches of upper Silver Bow as well as the canyon above Fairmont. The Opportunity site is representative of most of lower Silver Bow Creek and is similar to

most streambank deposits on the upper Clark Fork River as well.

All soil samples characterized by desorption analysis were grouped into 13 "type" textures (Table 5.2.1). The relationship between water content and water potential was fit to the soil hydraulic relationship developed by vanGenuchten [12] which also allows for prediction of relative unsaturated hydraulic conductivity [13]. Desorption and unsaturated hydraulic conductivity (K_u) curves for each "type" soil are presented in Figure 5.2.1 through 5.2.4.

$$S_e = \frac{1}{[1 + (\alpha h)^N]^{(1 - \frac{1}{N})}}$$

$$\text{where } S_e = \frac{(\theta - \theta_r)}{(\theta_s - \theta_r)} \quad [12]$$

θ = volumetric water content (cm^3/cm^3)

θ_r = residual water content

θ_s = saturated water content

N and α are vanGenuchten parameters

$$K_u = K_s S_e \left[(1 - (1 - S_e)^{\frac{1}{1 - \frac{1}{N}}})^2 \right] \quad [13]$$

where K_s = Saturated hydraulic conductivity ($\frac{\text{cm}}{\text{s}}$)

Table 5.2.1 Properties of "type" soils in the STARS investigation.

MATERIAL	CODE	BULK DENSITY (g/cm ³)	SATURATION θ_s (cm ³ /cm ³)	RESIDUAL θ_r (cm ³ /cm ³)	N ¹	α^1	K _s (cm/s)
TAILINGS MATERIALS							
SANDY TAILINGS	1	1.09	0.59	0.01	1.85	0.050	2.0*10 ⁻⁴
COARSE SAND TAILINGS	2	1.09	0.37	0.02	2.10	0.051	2.0*10 ⁻³
SILT TAILINGS	3	1.02	0.59	0.09	1.43	0.026	5.0*10 ⁻⁵
MIXED TAILINGS	4	1.07	0.60	0.19	1.31	0.047	2.0*10 ⁻⁴
SOIL MATERIAL							
SANDY SOIL	5	1.21	0.59	0.09	1.66	0.047	5.0*10 ⁻⁴
SANDY LOAM SOIL	6	1.21	0.60	0.06	1.64	0.029	5.0*10 ⁻⁴
SILT LOAM SOIL	7	1.02	0.62	0.16	1.33	0.018	1.0*10 ⁻⁴
SILTY SOIL	8	1.38	0.48	0.09	1.60	0.028	1.0*10 ⁻⁴
LOAM SOIL	9	1.1	0.59	0.12	1.50	0.030	2.0*10 ⁻⁴
SILTY CLAY SOIL	10	1.05	0.60	0.38	1.32	0.051	1.0*10 ⁻⁵
ORGANIC MUCK SOIL	11	0.83	0.69	0.30	1.34	0.028	1.0*10 ⁻⁴
COVERSOIL MATERIAL							
COVERSOIL	12	1.42	0.46	0.03	1.77	0.029	2.0*10 ⁻⁴

¹ The parameters N and α are fit by least-squares regression to the vanGenuchten equation which relates water content to water potential.

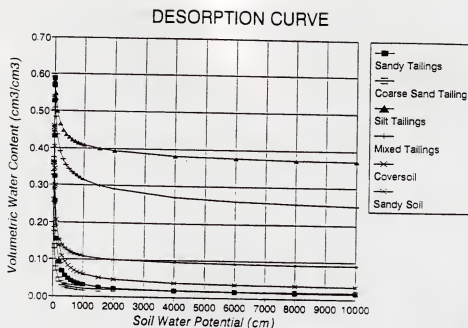


Figure 5.2.1. Desorption curves for type soils 1 through 6 for the STARS investigation.

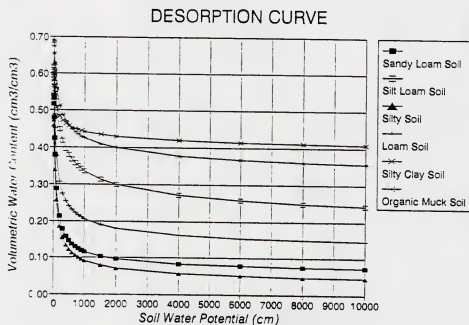


Figure 5.2.2. Desorption curves for type soils 7 through 12 for the STARS investigation.

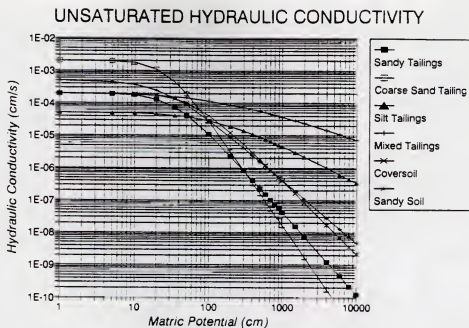


Figure 5.2.3. Unsaturated hydraulic conductivity curves for type soils 1 through 6 for the STARS investigation.

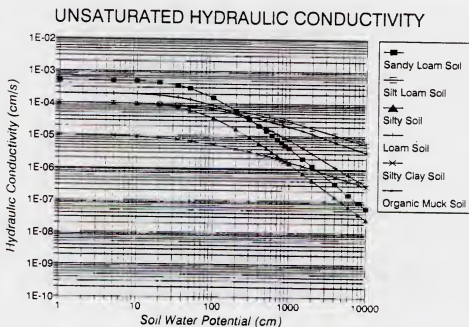


Figure 5.2.4. Unsaturated hydraulic conductivity curves for type soils 7 through 12 for the STARS investigation.

The domain modeled for each of the sites is depicted in Figure 5.2.5. The node and element numbering scheme corresponded to the convention used by other finite element models. Elements were thinner at the soil surface to avoid numerical instability problems associated with the steep water potential gradients that develop at the soil surface in response to evaporation. The type textures selected and the depth to the water table were based on field logs of lysimeter, piezometer, and neutron probe installations at each site.

The domain was modeled with a constant head boundary below the groundwater table to simulate the free water surface. The nodes at the soil surface were a special type called EI nodes within UNSAT2. During simulation, a potential evaporation or infiltration flux can be set. In an EI node, the actual evaporation or infiltration can be equal or less than the potential rate depending on the ability of the soil system to transmit water to satisfy the potential flux at the surface.

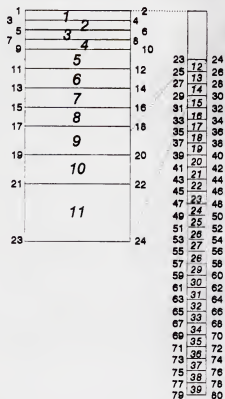
At each site, a typical climatic year was simulated in order to determine the quantity of groundwater recharge expected to occur in bare streamside tailings as well as in revegetated plots. In order to simulate the fully-drained soil profile (eg. field capacity), the domain was initialized as a nearly-saturated condition (a suction of -20 cm). A fixed head boundary of 0 cm of suction was set at the groundwater table at each site (Opportunity at 5.75 feet, Rocker at 4.5 feet, and Ramsay at 6.25 feet). The profiles were allowed to freely drain with no imposed surface flux for 90 days (180 days at the less permeable Ramsay site) to reach fully-drained conditions.

For the simulation, the year was partitioned into three time periods beginning on about June 15. The net water balance for each portion of the year was inferred from climatological observations. The first 90 days was assumed to have a net potential evaporation deficit of 25 cm which was applied equally throughout the 90 day "summer". The next 180 days consisted of a slow rate of surface infiltration amounting to 5 cm over the fall and winter. The final 90 days of the simulated year was the "spring" when precipitation exceeded evaporation. A total of 10 cm of infiltration was assumed to occur over the 90 day period. These values for potential surface flux were imposed equally for each day of the 0 to 90, 91 to 270, and 271 to 360 day simulation period. The head conditions existing at the end of the initial draindown period were used as input representing day 0 conditions.

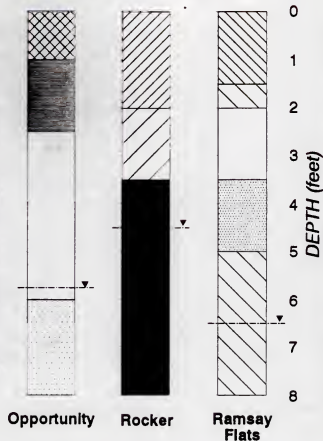
At each site, three individual soil surface conditions were simulated. A bare soil condition was used to simulate existing tailings. A vegetated plot representative of the deep plow plot was also simulated. Finally, because a small quantity of internal redistribution of water occurred after the initial 90-day drainage period, a zero surface flux condition (completely impermeable domain) was also simulated (Appendix E-2).

The model domain was changed from a fixed head boundary below the water table to a zero flux boundary during the year simulated. This head change was necessary to construct a mass water balance on the water lost to evaporation. Hence, if at the end of the

NODE AND ELEMENT NUMBERING SCHEME



SIMULATION DOMAIN



LEGEND

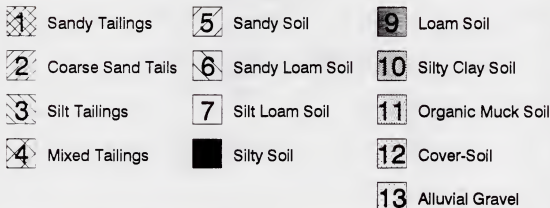


Figure 5.2.5.

Node and element numbering scheme and material property assignments for the UNSAT2 domain used for the STARS unsaturated flow simulation.

year the stored water of the bare soil or vegetated simulation was greater than the zero surface flux condition then groundwater recharge was expected to occur. Conversely, if either simulation was drier than the zero surface flux case, then no seepage to groundwater would be expected. The climate modeled was conservative in that slightly wetter conditions than the average observed from 1989 through 1992 was used. Hence these simulations would tend to slightly overestimate the tendency for streamside tailings to generate groundwater recharge in an average year.

Additional simulations were conducted at Rocker and Ramsay, the coarsest and finest-textured tailings areas. The effects of capillary rise of water from groundwater and its contribution to surface evaporation rate is particularly important in determining the overall direction of flux in streambank soils. When groundwater is close enough to the soil surface to be "pumped" to the surface by evaporation, a net upward gradient develops and the groundwater system recharges the vadose zone except for infrequent drainage events in spring. Simulations were conducted to investigate the potential contribution of groundwater at variable depths to the surface evaporation rate. Potential evaporation from bare soil corresponding to average flux (25 cm in 90 days) was simulated at Rocker and Ramsay. In addition, rapid evaporation (0.6 cm/day) was also simulated at Ramsay.

5.2.3 UNSATURATED FLOW SIMULATION

Detailed output as well as input files for each simulation are contained in Appendix E-2. Descriptions of the unsaturated flow simulations, depiction of the water content and suction profiles in each soil, and an overall annual water balance is presented for each site in the subsequent section.

Rocker: Change in stored water under the three modeling scenarios is presented in Figure 5.2.6. The water content (Figure 5.2.7) of the zero flux soil profile changed imperceptibly over the year due to rapid equilibration of the system to the water table influence during the initial draindown period. The influence of the shallow water table on soil water content as well as matric potential is evident (Figure 5.2.8). Throughout the domain, the matric potential (h) was nearly equal in magnitude to the distance above the water table (z) so that total head throughout the domain was essentially zero. The matric potential of the zero flux case was sufficient to reduce the water content of the coarse-textured materials prevalent at Rocker to less than 10 percent (volumetric) in the tailings layer (top 2 feet). As a consequence of the static conditions in this simulation, the sum of stored soil water in the profile was constant throughout the simulation (Figure 5.2.6)

In the bare soil evaporation case, a measurable quantity of water was withdrawn from the coarse-textured tailings in the upper 2 feet of the soil profile (Figure 5.2.9). The matric potential (Figure 5.2.10) had decreased to -15,000 cm (-15 bars, the imposed value for minimum permissible head at the surface node) at the soil surface in response to summer evaporation. This decline in matric potential created a very steep upward gradient causing

water to move to the surface from a depth of about 1 to 2 feet. The rate of water movement in the surface layer became so slow, however, that only 4.2 cm of actual evaporation occurred (Figure 5.2.6), a small fraction of the potential evaporation of 25 cm. During the subsequent fall and winter period, the imposed infiltration of 5 cm more than replenished the water withdrawn by evaporation so that water moved downward through the profile (Day 270). Additional percolation occurred during the rapid infiltration "spring" period (Day 360). Overall, the bare soil at Rocker had a net groundwater recharge of 8.4 cm or about 25 percent of the average annual precipitation.

Simulation of a revegetated plot at Rocker suggested that plant roots would remove water to a much greater depth than surface evaporation. A significant decrease in water content (Figure 5.2.11) occurred throughout the domain, but the decline in matric potential (Figure 5.2.12) was most pronounced within the root zone (above 3 feet). Despite the fact that only a few percent of the root biomass in this simulation existed below the upper foot of soil, a significant quantity of water was withdrawn between 1 and 3 feet in depth.

Overall, the bare streambank tailings areas would be expected to have 8.4 cm of percolation over the year while the revegetated plot (Figure 5.2.6) was not expected to generate any percolation. The revegetated plot had 5.2 cm less water at the end of the year than at the beginning. Vegetation at the Rocker site was expected to be sub-irrigated as evidenced by the gradual decline in the water table during the summer (Figure 5.2.12, Day 90).

A second simulation was conducted at the Rocker site to identify the critical groundwater depth at which significant capillary rise could occur. The shallow groundwater was shown to benefit established vegetation through subirrigation. Capillary rise was considered to be significant for the bare tailings system when an accelerated rate of surface evaporation could be supported by upward capillary flux from the water table. The water content (Figure 5.2.13) and matric potential (Figure 5.2.14) are shown for the Rocker site with groundwater fixed at 2, 3, and 4 feet in depth. Surface evaporation was equal to potential evaporation (imposed potential evaporation was 0.278 cm/day) when the water table was either 2 or 3 feet from the surface. If the water table was deeper than 3 feet, no measurable flux from groundwater occurred.

Rocker Unsat Model

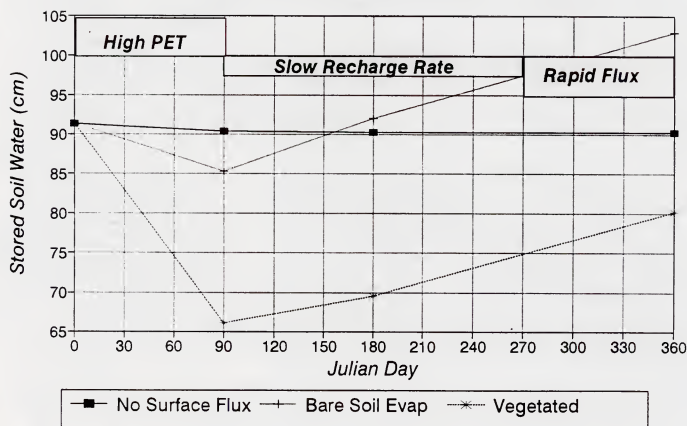


Figure 5.2.6. Predicted changes in stored soil water for the Rocker simulation.

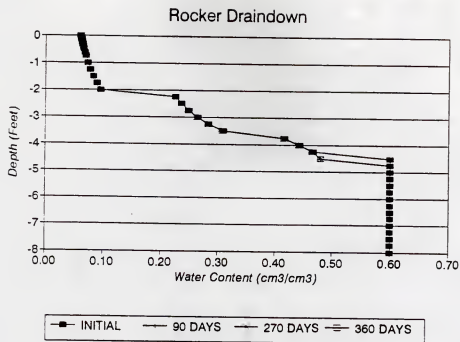


Figure 5.2.7. Soil water content after 0, 90, 270 and 360 days of simulation for the zero surface flux case at Rocker.

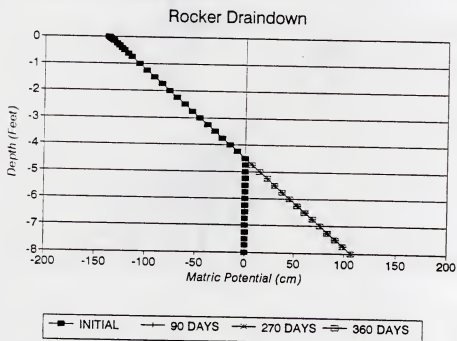


Figure 5.2.8. Soil suction after 0, 90, 270 and 360 days of simulation for the zero surface flux case at Rocker.

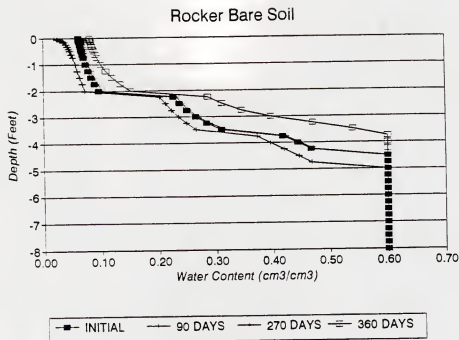


Figure 5.2.9. Soil water content after 0, 90, 270 and 360 days of simulation for the bare soil evaporation case at Rocker.

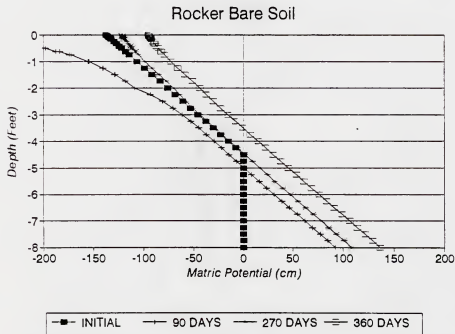


Figure 5.2.10. Soil suction after 0, 90, 270 and 360 days of simulation for the bare soil evaporation case at Rocker.

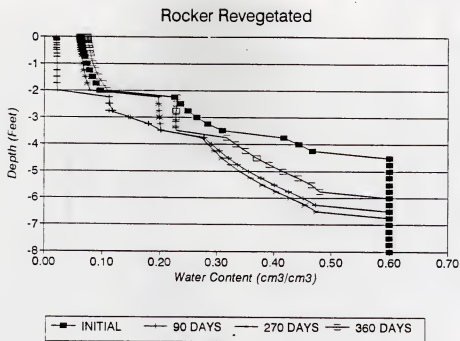


Figure 5.2.11.

Soil water content after 0, 90, 270 and 360 days of simulation for the revegetated case at Rocker.

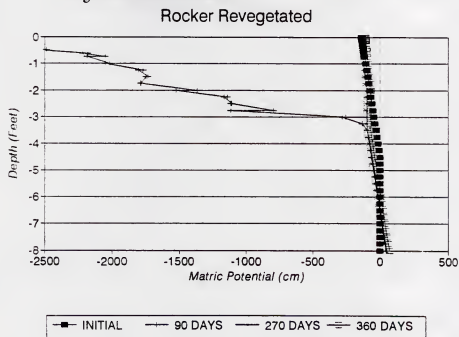


Figure 5.2.12.

Soil suction after 0, 90, 270 and 360 days of simulation for the revegetated case at Rocker.

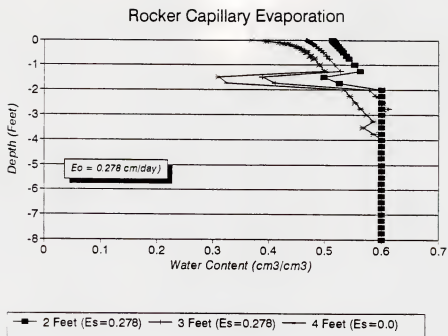


Figure 5.2.13. Soil water content for a fixed water table at 2, 3, and 4 at the Rocker site. Groundwater within 3 feet of surface contributed to evaporation.

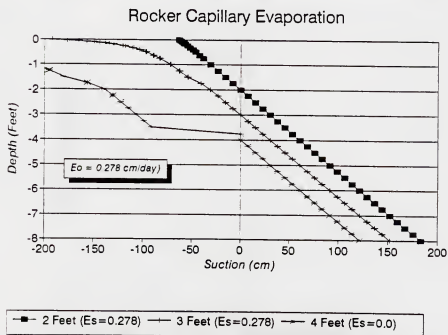


Figure 5.2.14. Soil suction for a fixed water table at 2, 3, and 4 at the Rocker site. Groundwater within 3 feet of surface contributed to evaporation.

Ramsay Flats: Patterns of water use and changes in stored soil water simulated at the Ramsay Flats site were similar in many regards to Rocker despite significant differences between the sites (Figure 5.2.15). Ramsay soil had a slower hydraulic conductivity and greater depth to groundwater than at Rocker. The water content and matric potential of the zero surface flux case (Figure 5.2.16 and 5.2.17) changed gradually over the period simulated due to the restricted rate of flow through the silty clay-textured soil layer between 3.5 and 5 feet in depth ($K_s = 5 \cdot 10^{-5} \text{ cm/s}$). Vertical variations in water content (Figure 5.2.16) between 1.5 and 2 feet and between 5 and 6.25 feet were due to textural stratifications.

In the bare soil case, evaporation decreased the water content (Figure 5.2.18) and the matric potential (Figure 5.2.19) to a depth of 3 feet. A total of 6 cm was removed by evaporation during the summer period. Surface evaporation does not normally affect water content much below the top foot of soils. At the Ramsay site, drying may have occurred to a greater depth due to the ability of the silty soil to continue to transmit at fairly low matric potential. Surprisingly, silt and clay-textured soils have a greater hydraulic conductivity than sands at low matric potential. In Figure 5.2.3 for example the silt tailings had a saturated K_s nearly 1000 times slower than coarse sandy tailings. Their unsaturated K_u values at a suction of 100 cm were equal while at 1000 cm of suction the silt had a K_u that was 500 times faster than the sand. Overall for the year (Figure 5.2.15) the bare soil site gained 7.2 cm of water, only slightly less than at the coarser-textured Rocker site. Despite the slow permeability of the tailings at Ramsay, an appreciable amount of groundwater recharge is expected where bare tailings are exposed.

The simulation of the vegetated plot suggested that water would be removed by plants to the top of the silty clay layer at 3.5 feet in depth (Figure 5.2.20). Plants withdrew 19.7 cm of water during the summer period. The matric potential of the vegetated Ramsay plot (Figure 5.2.21) dropped to -15 bars at the soil surface and ranged from -2 to -6 bars in the remainder of the root zone. Recharge the following winter created an obvious wetting front as water in the upper 1.5 feet of tailings was replenished. After the spring recharge period, the vegetated case at Ramsay was 5.8 cm drier than at the beginning of the year indicating that groundwater recharge is not likely to occur after vegetation becomes established.

The pattern of water withdrawal by plants during the 90 day summer period is shown in more detail in Figure 5.2.22 and 5.2.23. Plants initially drew much of their water from the upper part of the root zone. As the soil surface became drier, a zone with steep matric potential gradient formed. As time progressed, the steep gradient region migrated deeper into the soil profile as increasing amounts of soil water were utilized from the base of the root zone.

Simulation of the capillary influence of the groundwater table at Ramsay was nearly identical to the results at Rocker (Figure 5.2.24 and 5.2.25). When groundwater was within 3 feet of the soil surface then upward flux occurred from groundwater into the vadose zone. Groundwater at four feet and deeper had no effect on surface evaporation.

Ramsay Flats Unsat Model

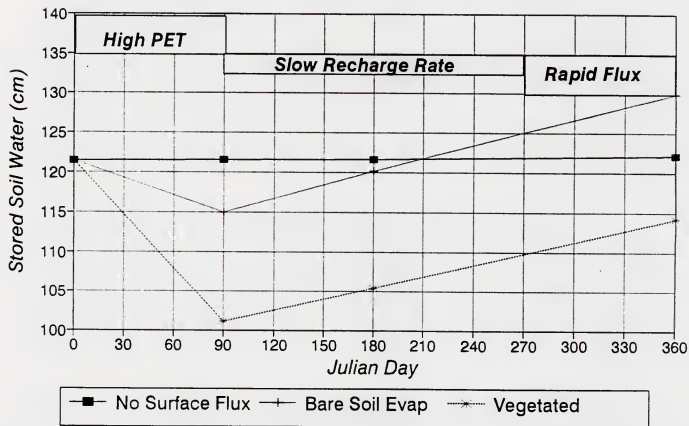


Figure 5.2.15. Predicted changes in stored soil water for the Ramsay simulation.

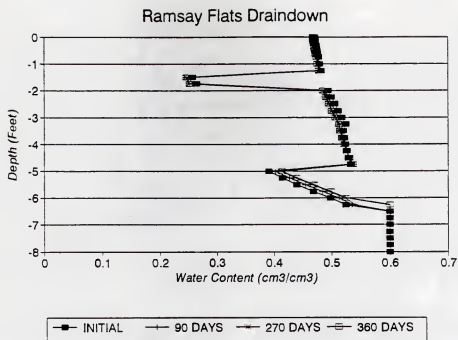


Figure 5.2.16. Soil water content after 0, 90, 270 and 360 days of simulation for the zero surface flux case at Ramsay.

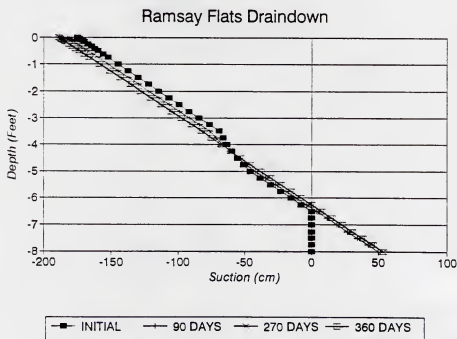


Figure 5.2.17. Soil suction after 0, 90, 270 and 360 days of simulation for the zero surface flux case at Ramsay.

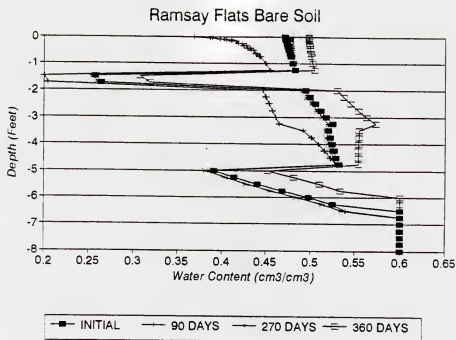


Figure 5.2.18. Soil water content after 0, 90, 270 and 360 days of simulation for the bare soil evaporation case at Ramsay.

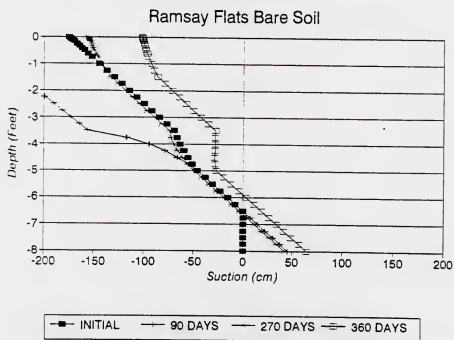


Figure 5.2.19. Soil suction after 0, 90, 270 and 360 days of simulation for the bare soil evaporation case at Ramsay.

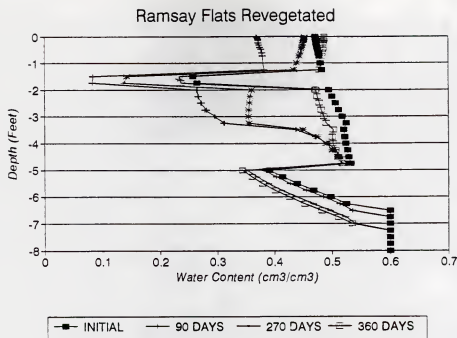


Figure 5.2.20. Soil water content after 0, 90, 270 and 360 days of simulation for the revegetated case at Ramsay.

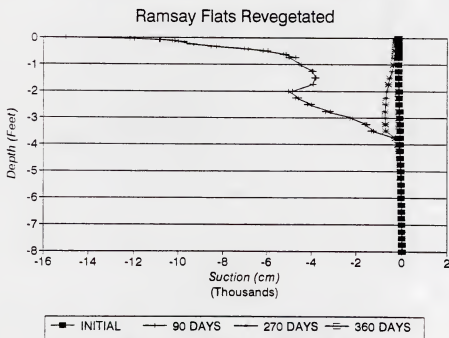


Figure 5.2.21. Soil suction after 0, 90, 270 and 360 days of simulation for the revegetated case at Ramsay.

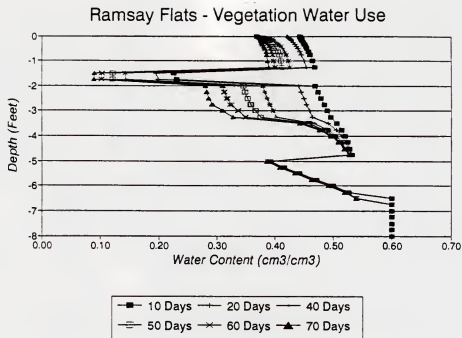


Figure 5.2.22. Simulated soil water content after intervals of plant water removal by vegetation at Ramsay.

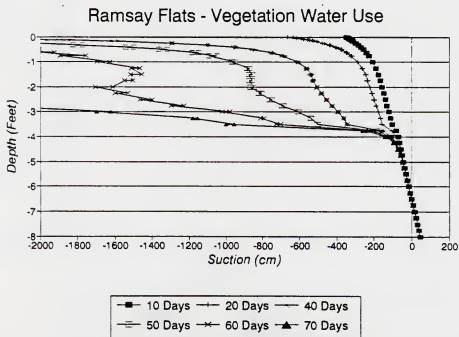


Figure 5.2.23. Simulated soil suction after intervals of plant water removal by vegetation at Ramsay.

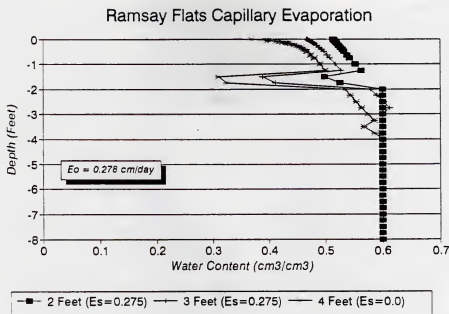


Figure 5.2.24. Soil water content for a fixed water table at 2, 3, and 4 at the Ramsay site. Groundwater within 3 feet of surface contributed to evaporation.

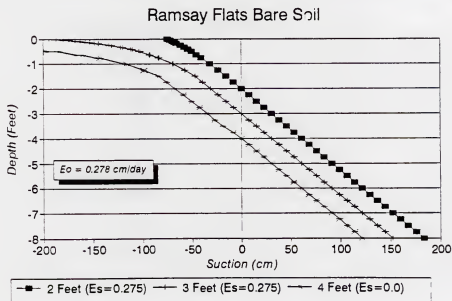


Figure 5.2.25. Soil suction for a fixed water table at 2, 3, and 4 at the Ramsay site. Groundwater within 3 feet of surface contributed to evaporation.

Opportunity: The Opportunity site has soil properties that are intermediate between Rocker and Ramsay which are apparent in changes of stored soil water through the year (Figure 5.2.26). Groundwater at Opportunity was 5.75 feet below ground surface. The zero surface flux case at Opportunity indicated that internal drainage continued to occur within the model domain during the simulation period (Figure 5.2.27 and 5.2.28). The apparent loss of water within the domain was an artifact of the method used to calculate water content from reported values for matric potential. The vanGenuchten equation is only continuous across negative values for head, consequently erroneous values were computed by the UNSAT2 code below the water table (at 6 to 7 feet). Nonetheless, these computation errors do not affect the comparison of results between the zero flux case, the bare soil, and the revegetated condition.

In the bare soil plot at Opportunity, 1.7 cm of evaporation (Figure 5.2.26) occurred in the upper 1 foot of soil (Figure 5.2.29). Subsequent recharge of the soil occurred so that overall the bare soil plot gained 9 cm of water relative to the zero surface flux case. The matric potential curves for the bare soil case (Figure 5.2.30) indicated that the entire soil profile to the groundwater table was recharged by infiltration in the spring.

Vegetation in the Opportunity simulation withdrew water to a depth of nearly 4 feet (Figure 5.2.31) but the majority of water was removed from the upper 2.5 feet of soil. The matric potential curves (Figure 5.2.32) clearly indicate that vegetation induced water to move upward from a depth of 5 feet during the summer. During the subsequent spring and summer, the profile was nearly recharged. Overall, vegetation had little effect on stored soil water at Opportunity indicating that groundwater recharge is not likely once vegetation is established.

Opportunity Unsat Model

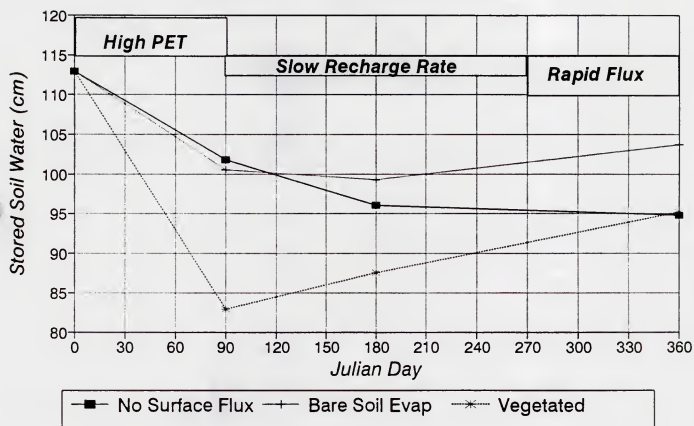


Figure 5.2.26. Predicted changes in stored soil water for the Opportunity simulation.

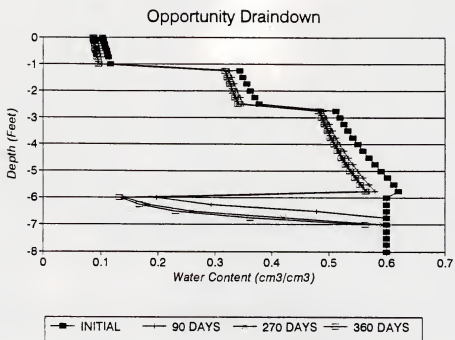


Figure 5.2.27. Soil water content after 0, 90, 270 and 360 days of simulation for the zero surface flux case at Opportunity.

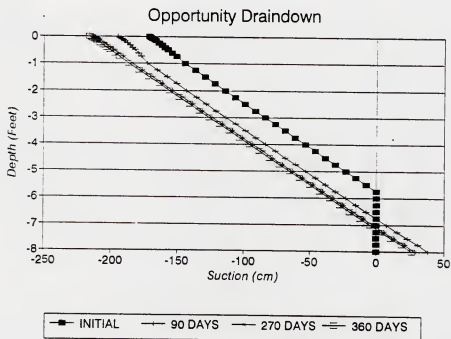


Figure 5.2.28. Soil suction after 0, 90, 270 and 360 days of simulation for the zero surface flux case at Opportunity.

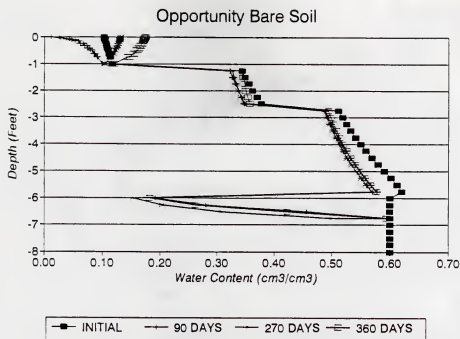


Figure 5.2.2° Soil water content after 0, 90, 270 and 360 days of simulation for the bare soil evaporation case at Opportunity.

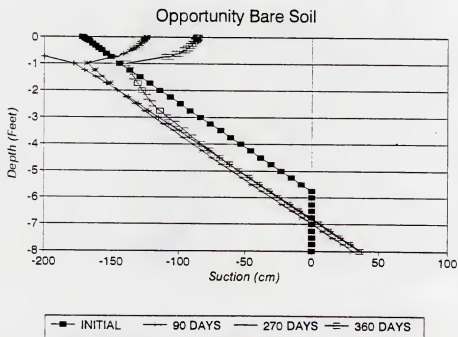


Figure 5.2.30. Soil suction after 0, 90, 270 and 360 days of simulation for the bare soil evaporation case at Opportunity.

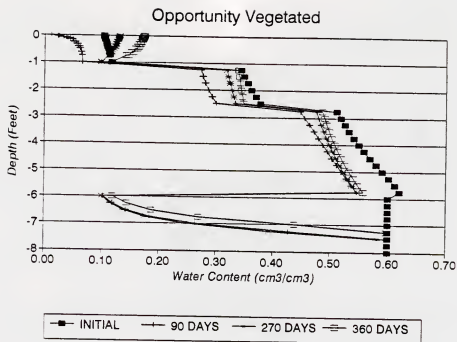


Figure 5.2.31. Soil water content after 0, 90, 270 and 360 days of simulation for the revegetated case at Opportunity.

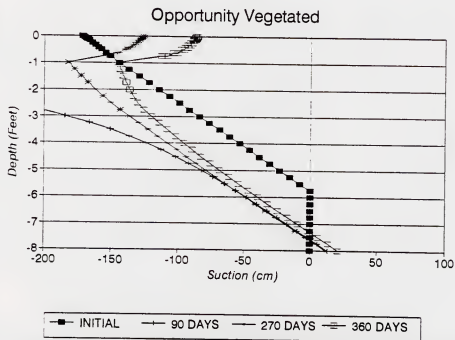


Figure 5.2.32. Soil suction after 0, 90, 270 and 360 days of simulation for the revegetated case at Opportunity.

5.3 GEOCHEMICAL MODELING

The transport of metals from tailings through the vadose zone presents a potential threat of contamination to groundwater. To describe the transport of metals through the vadose zone and evaluate the potential for groundwater contamination it is necessary to understand both the soil hydrologic regime (section 5.2) and geochemical processes occurring in the vadose zone. Factors known to be important in controlling metal mobility in the vadose zone include pH, Eh, adsorption and precipitation reactions (Dragun 1988). In addition, the speciation of dissolved metal plays a major role in metal behavior, bioavailability and relative toxicity.

Metals in soil/tailings systems are present in several forms or "compartments" including, from greatest to least reactive:

soluble > exchangeable > adsorbed = co-precipitated = precipitated

The activity of metals in the soluble form is a function of ion size and charge and the overall ionic strength of the solution. Relative to other forms of metals, soluble metals are very reactive. Exchangeable metals are bound to soil surfaces by electrostatic forces (ionic). Exchange reactions are readily reversible and depend upon the amount of exchangeable metals on surfaces and in solution. Adsorbed metals are bound into the structure of the adsorbing surface, typically iron, aluminum and manganese hydroxides and oxides, via the sharing of a bond (covalent) with a ligand on the adsorbent surface. Adsorbed metals are thus held more strongly than exchangeable metals. Metals which are co-precipitated or precipitated are somewhat less reactive and available for plant uptake than those which are adsorbed and are only released slowly to solution to maintain equilibrium as soluble levels decrease.

Geochemical modeling can be used to gain an understanding of the factors affecting metal behavior and the speciation of the dissolved metal fraction.

5.3.1 Background and Model Selection

Geochemical modeling was performed on several pore water samples collected from lysimeters during 1990 and 1992 using the EPA chemical speciation model MINTEQA2 (EPA 1991), a public domain Fortran code. MINTEQA2 utilizes a large thermodynamic database and a simple mass balance approach and the phase rule to speciate input constituents in dissolved, adsorbed and solid phases utilizing the Newton-Raphson iteration procedure. Kinetic processes in the model are not simulated and care must be taken in the interpretation of model results to consider the role of kinetics, especially in the formation of various solid phase precipitates.

MINTEQA2 uses an interactive program PRODEFA2 to create input files. Analytical data is entered into PRODEFA2 (EPA 1991) as "components", which are the basic building-blocks for all species (dissolved, adsorbed or solid) in the system which can be formed. For example Ca^{2+} and CO_3^{2-} are both components which can combine to form a soluble species (eg: CaCO_3^0), or a solid (calcite (CaCO_3)). Likewise Ca^{2+} can combine with any other thermodynamically allowable component. The components are partitioned to various soluble species and/or solids and represent an accounting system by which mass balance is maintained.

MINTEQA2 utilizes six "Types" of species (Type I through Type VI) of components and species. Type I contains the components which are species in solution (eg: Ca^{2+}). Values for molarity (concentration) and activity are output. Type II species are all other soluble ion-pairs or adsorbed complexes (eg: $\text{Ca}(\text{OH})^+$) and may be considered as aqueous and adsorption reaction products. Type III species are those with a fixed activity. These are species that are present at a fixed equilibrium activity and can include solids, soluble species or dissolved gases. The user may define several parameters as fixed (eg: pH or Eh) or other components.

Type IV species are finite solids. These are solids that are presumed to be present (user entered) or to precipitate from the solution. Finite solids are allowed to dissolve and re-precipitate in accordance with thermodynamic laws. Type V species are possible (undersaturated) solids. These are solid phases that are defined by MINTEQA2 as relevant to the components in the input file but are undersaturated and do not precipitate. If oversaturation and precipitation do occur, the precipitate is then listed as a Type IV finite solid. Type VI species are those components and solid phases which are either automatically or explicitly excluded from the mass balance calculations. For example, a solid phase which may form from a solution based on thermodynamic data but is known to be kinetically prohibited can be excluded in favor of a more realistic solid phase.

Another feature of MINTEQA2 which was useful for pore water modeling included a "sweep" option. The sweep option allows one to vary the total concentration or fixed activity of one component at specified increments while maintaining the remainder of the input intact. For example, a sweep of pH can be run from 3 to 8 at one unit increments on a solution. The output from the previous sweep is used as input for the next sweep. This option was useful for evaluating the effect of liming (and attendant increase in pH) on metal behavior. The reader is referred to the User's Manual for a more detailed discussion of MINTEQA2 (EPA 1991).

In addition to MINTEQA2, the model GEOCHEM (Sposito and Mattigod 1979, Parker et al 1987) was employed to compare with the results of arsenic adsorption modeling in MINTEQA2. GEOCHEM is a Fortran code which is extremely similar to MINTEQA2. It employs a similar thermodynamic database and the use of mass balance and was developed from the same parent model (REDEQL). However, GEOCHEM has a more extensive thermodynamic database for arsenic speciation.

5.3.2 Modeling Approach

Pore water solutions modeled were selected to represent a variety of geochemical conditions (eg: varying pH and metal concentrations). It was difficult to use the model for a comparison of treatment effectiveness due to incomplete lysimeter sample recovery, the heterogeneity of the tailings material, and the lack of replicated data. Where analyte concentrations were reported below the detection level, the detection level was used as an input value. This was most prevalent for 1990 PO_4 data and would have the greatest effect on Mn solubility at low and neutral pH values due to the formation of MnHPO_4 solid in many of the model runs.

The primary objectives of the modeling were 1) to determine the distribution of ion species in pore water in several geochemical environments (low to neutral pH) and; 2) determine what solid phases may be forming or controlling the metal/ligand chemistry. The effects of sorption on arsenic behavior and the effect of added lime on the geochemical behavior of constituents were also evaluated. Modeling was not intended to evaluate the performance of individual treatments and caution should be exercised when using modeling results to compare treatments. It is better to utilize actual field data to compare treatment effectiveness.

Four general modeling approaches were taken. The first approach examined soluble metal speciation and potential solid phase controls on metals. In this approach metal and major cation and anion concentrations measured in selected suction lysimeters were input to the model, along with alkalinity and pH values. The measured value for pH was fixed. Because of the ambiguity of field Eh measurements, Eh was not entered into the input file, but was allowed to be calculated by the model. Additionally, no Eh sensitivity analysis was performed but the effects of Eh on metal solubility are discussed at the end of this section. The fixed pH and alkalinity values determined the carbonate equilibrium of the system. When pH was less than 4.5, no alkalinity values were entered. Instead, the partial pressure of CO_2 ($p\text{CO}_2$) was fixed at 0.003 atm and carbonate speciation was determined by MINTEQA2. The value of 0.003 atm is equal to a CO_2 partial pressure $10^{-2.5}$ moles/l, which is 10 times greater than atmospheric CO_2 levels and representative of levels commonly found in soils (Lindsay 1979). No solid species were allowed to precipitate, but saturation indices were calculated to determine what solid phases were thermodynamically favored. A saturation index is the ion activity product for a given species. A positive value indicates oversaturation. Values are in log units. It is worth noting that speciation of metals does not account for metals that are, under actual field conditions, adsorbed. This predicted solution metal concentrations may be greater than actual field values.

The second approach to modeling examined the effect of incremental increases in pH on metal behavior. This modeling simulates the effects of excess precipitation leaching through the tailings and into the underlying soil. In this case, the buried soil acts as a buffer, exerting a control on the pH of the solution, hence the model is analogous to a pH buffer curve. Through time, the buffering capacity of the soil will be exhausted and pH will

decrease vertically through the profile. Although the buffering effect of the soil cannot be quantified in terms of capacity, pore water data do show that the pH in the majority of the underlying soils is near neutral after approximately 80 years of leaching by tailings solutions. The $p\text{CO}_2$ was fixed (0.003 atm) and solids were allowed to precipitate. The solids Al_2O_3 , diaspore, hematite and goethite were excluded in these runs because more soluble hydroxide species are kinetically favored during initial precipitation reactions involving Al and Fe. Additionally, it was necessary to preclude the precipitation of other solid phases to achieve convergence for some sample runs, presumably due to the complexity of the system and large number of input parameters. All excluded solids are listed in Appendix E-3 for specific model runs.

The third approach attempted to evaluate the effect of direct limestone addition on metal behavior and to identify pH buffering plateaus by incrementally adding CaCO_3 (as aqueous components Ca and CO_3) to the sample. The incremental addition of CaCO_3 would also simulate the field conditions of partial mixing of lime amendment as well as the potential effects of underliming on metal behavior. Output analyte concentrations from a previous run were used as input for successive runs with the addition of more CaCO_3 . The $p\text{CO}_2$ was fixed (0.003 atm) and pH was calculated by the model. Precipitation of solids was allowed. This approach was attempted on several pore water samples but the model proved not numerically robust enough to achieve convergence. The lack of convergence was probably due to the number of input parameters and the complexity of the multiple solid phase equilibrium reactions, and phase rule violations due formation of multiple solid phases. This approach was successful, however, using a saturated paste extract collected from a control plot (0 - 8") at Ramsay Flats (site 21).

Finally, arsenic adsorption was modeled by using a range of selected synthetic As concentrations in the presence of ferric oxide (goethite) as an adsorbent, which together with iron hydroxide (ferrihydrite) are likely present in substantial quantities in tailings and soils. Ferric oxide was chosen because of the availability of reliable thermodynamic and because it has a lower adsorptive capacity and is thus a more conservative choice than ferric hydroxide. The constant capacitance adsorption model in MINTEQA2 was used. The characteristics of the adsorbent were varied to evaluate the effect on As sorption. Values for the adsorbent were taken from the literature (Goldberg 1986). Additionally, modeling was performed on several pore water samples (Table 5.3.1) to calculate equilibrium arsenic concentrations for use as a comparison to actual pore water results and the solubility of ferric and calcium arsenates. Arsenic modeling was performed both with and without other cations present. Additionally, because it contains a more comprehensive As thermodynamic database, the model GEOCHEM (Sposito and Mattigod 1979) was utilized to compare As speciation against results obtained from MINTEQA2. This approach was taken for the modeling of arsenic because of the documented role of adsorption in controlling arsenic solubility (Lindsay 1979, Bohn et al 1985). All modeling was performed at 25° C.

5.3.3 Results

Results for all modeling efforts are presented in Appendix E-3. They are presented in order of discussion with respect to modeling approach (*eg*: fixed pH output first) and within this order by site number and sample depth). All results include input parameters, percent distribution of dissolved ions, excluded solids (not permitted to precipitate), and saturation indices. The results for the pH sweeps and CaCO_3 addition also present solid phases (finite solids) present at equilibrium.

It should be pointed out that when an elemental symbol for a metal is used in the text (*eg*: Al) it refers to the total dissolved concentration of Al, including all ionic form or valence states. When a specific ion species is discussed the valence is shown (*eg*: Al^{3+}). The use of SO_4 , PO_4 , CO_3 , NO_3 , F and Cl also refer to the ion in general terms. This convention is used because all analytical results were for total dissolved concentration of the analyte and no speciation was done. All input components, however, have specific valence states (*eg*: Al^{3+} , SO_4^{2-}) and the reader is referred to Appendix E-3 for a listing of input components for each model run.

Table 5.3.1. Pore water and saturated paste samples used in geochemical modeling.

SITE	TREATMENT	DEPTH (cm)	SAMPLE ¹	FIXED pH	pH SWEEP	CaCO ₃ ADDITION	ARSENIC CHEMISTRY ²	
							MINTEQA2 ³	GEOCHEM
33	control	40	9233C40	X	X		X	X
33	ag-till	40	9033A40	X				
21	control	90	9221C90	X				
33	control	90	9233C90	X	X			
07	control	40	9007C40	X	X			
21	control	40	9007C40	X	X			
07	ag-till	90	9207a90	X			X	X
07	deep plow	150	9207d150				X	X
21	injection	150	9221i150				X	X
33	topsoil	40	9233t40				X	X
07	control	90	9207c90					X
07	control	150	9207c150					X
33	ag-till	90	9233a90					X
21	injection	90	9221i90					X
33	topsoil	150	9233t150					X
21	control	0 - 8 ⁴	9021c ⁴			X		

¹ - first 2 numbers in sample represent year 19-- of collection.

² - pore water speciation data obtained using GEOCHEM;
adsorption data obtained using MINTEQA2

³ - synthetic arsenic adsorption runs not listed

⁴ - saturated paste extract sample from 0 to 8 inches.

Fixed pH Modelling: A pore water sample (33c40) collected in 1992 from the 40 cm depth of the control plot at site 33 (Opportunity) was modeled to evaluate metal speciation at low pH. The lysimeter was installed in tailings material. Solution pH was 3.96 and represents the low end of the pH values observed during pore water sampling. Modeling input and output parameters are presented in Appendix E-3.

Modeling results indicate that the majority of soluble metals were either free or paired with SO_4 . Exceptions to this were Al, 40 % of which was speciated with F and 40 % with SO_4 , and Fe, 75 % of which was paired with hydroxyl groups (OH). A large percentage of Cd (48 %), Cu (63 %), Mn (65 %) and Zn (56 %) were present as free ions. Of the alkaline metals, greater than 94 % of Na and K were present as free ions and approximately 65 % of both Ca and Mg. The remainder of the Ca and Mg was present as SO_4 ion pairs.

The majority of anions were present in the free anion form, including SO_4 (63 %). Phosphate and As were present predominantly as protonated species, with about 10 % of the PO_4 paired with Ca and Mg. All of the NO_3 was unpaired. The only anion besides SO_4 that had an appreciable percentage of its total dissolved concentration paired with a metal was F, with greater than 99 % paired with Al.

A review of the saturation indices reveals that several precipitates for specific metals were thermodynamically favored. However, formulation of many of these solids are known to be limited by kinetic processes. For example, hematite and diasporite are low solubility, stable oxides of Fe and Al, respectively, and are known to form either at high temperature or after more soluble Fe and Al hydroxyoxides (eg: gibbsite and ferrihydrite) form and stabilize. Iron was oversaturated with respect to several low energy hydroxides and both Fe (jarosite) and Al (alunite) hydroxy-sulfates were present. Iron also formed a coprecipitate with Cu (cupric ferrite). The most obvious results of the saturation indices calculations were the lack of formation of any precipitates for Pb, Cd, Mn, Zn and As.

To evaluate the geochemistry of a worst case tailings pore water, a 1992 pore water sample from the 90 cm depth on the ag-till plot at site 7 (9207a90) was modeled. This sample had the lowest pH (2.69) and highest metal and SO_4 concentrations of any samples collected during the study.

Results were very similar to 9233c40 in that the majority of metals and major cations were present as either free ions or paired with SO_4 . Aluminum showed a strong affinity for F, with 97 % of F complexed with Al. Only 14 % of Fe was complexed with hydroxyl groups and 99 % of PO_4 was complexed with Fe, due to the very high concentrations of Fe (1740 mg/l input) relative to all other samples. Only Fe hydroxyoxides and hydroxysulfates (jarosite) and Al hydroxysulfate were present as oversaturated ion pairs.

To examine the effect of amendment incorporation on pore water geochemistry, a pore water sample collected from the ag-till plot at site 33 in 1990 at the 40 cm depth was modeled. The lysimeter was installed in tailings material approximately 10 cm below the

depth of amendment incorporation. No sample was collected for metals analyses in 1992 due to low sample volume. However, pH and SC sample readings were obtained and indicate that the pH was similar to that observed in 1990 (5.00 vs. 5.57 and 6.49 vs 6.36 (SC), respectively). It is therefore likely that the metal solution composition was similar in 1990 and 1992. This is supported by observations from all sights at various depths that show little change in pore water chemistry through time after amendment incorporation.

The results indicate that the majority of metals and alkaline earth cations were paired with HCO_3 or are present as free ions. The exceptions to this are Al, which was 100 percent complexed with F, and Fe, which was completely paired with hydroxyl groups. Copper (47 %) and Pb (58 %) were predominantly paired with carbonate and bicarbonate, but all had a substantial percentage of total dissolved concentration present as free metal (Cu = 48 %, Pb = 32 %). Manganese (89 %), Cd (72 %), and Zn (75 %) were mostly in the free ionic form, as were Ca (89 %) and Mg (90 %). Greater than 99 % of K and Na were also in the free ionic form.

All Cl and NO_3 were unpaired. Seventy-two percent of PO_4 was protonated, with the remainder complexed with either Ca or Mg. As was the case with the control sample, the majority of the F (60 %) was paired with Al. Most of the SO_4 was also free (55 %), with the majority of the remainder paired with Ca and Mg.

The increase in pH due to lime addition resulted in a higher concentration of bicarbonate and hydroxyl ions relative to SO_4 . Thus more of the Fe was complexed with hydroxyls and Pb and Cu were paired with bicarbonate and carbonate. More metals, however, including Mn, Cd, Zn, were unpaired. This may be in part due to a considerable lack of charge balance in the sample, where the cation to anion difference was 75 %, possibly due to an artificially low analytical SO_4 level.

The saturation indices indicate that Fe and Al were present as iron hydroxides and alunite, respectively. These precipitates were also observed in the control sample. However, the increase in pH as a result of amendment incorporation did result in the precipitation of Cu (as malachite ($\text{Cu}_2(\text{OH})_2\text{CO}_3$) and azurite ($\text{Cu}_3(\text{OH})_2(\text{CO}_3)_2$) and octavite (CdCO_3).

A sample collected in 1992 from the 90 cm depth of the control plot was modeled at site 21 (Ramsay Flats) to evaluate metal speciation at a neutral pH. The lysimeter was installed in topsoil underlying tailings. Solution pH was 7.12 and represents the neutral pH values typically observed in pore water samples from buried topsoil. Modeling input and output parameters are presented in Appendix E-3.

Although the results from this model run are for a buried soil (versus tailings) material collected from a different site (21) than the sample discussed above (site 33), it can serve as a comparison between metal behavior at a low versus neutral pore water pH. Because of the higher pH value, bicarbonate (HCO_3) and hydroxyl (OH) ions were present in higher concentrations and played a more significant role in ion pairing, decreasing the

importance of SO_4 as the dominant anion in ion pairs. Generally, the percentage of a given metal present as a free ion decreased at neutral pH values and more metals were paired with HCO_3 and OH . However, a considerable percentage of Cd, (43 %), Mn (68 %) and Zn (48 %) remained present as an unpaired ion, showing little change from a low pH environment. The amount of Fe paired with OH increased to 98 % and 17 % of the Cu was present as a hydroxyl pair. Contrary to a low pH environment, where a large percentage of the Al was paired with F, the majority of Al (90 %) was paired with hydroxyl groups, even though the input concentration of F was the same as for the lower pH model run. The majority of Cu (55 %) and Pb (80 %) were paired with CO_3 versus SO_4 at lower pH values. Although SO_4 concentrations were lower at a neutral pH, approximately 30 % of Cd, Mn, Zn as well as Ca and Mg, still formed ion pairs with SO_4 . The remainder of Ca and Mg were present as free ions and their ion pairing characteristics changed little with an increase in pH. The same was true for K and Na.

Anion behavior did not change significantly, with 100 % of NO_3 and Cl, and 70 % of SO_4 present as free ions. The majority of F was present as a free ion (87 %), with the remainder bound with either Ca and Mg, compared to virtually all of the F paired with Al at lower pH values. Approximately half of the PO_4 was protonated and half paired with either Ca or Mg.

The biggest change in possible precipitates as indicated by the saturation indices was the presence of various Al hydroxides. Alunite was also present. Copper was also oversaturated at neutral pH with respect to cupric ferrite ($\alpha\text{-CuFe}_2\text{O}_4$) and malachite ($\text{Cu}_2(\text{OH})_2\text{CO}_3$), a copper hydroxy-carbonate. The solubilities of these minerals are similar and it is possible that both are present. Although cupric ferrite is slightly less soluble than malachite, malachite (or a similar-looking mineral) has been observed at many of the study sites. The solution was also slightly oversaturated with respect to calcite, gypsum and dolomite. Iron hydroxides remained the dominant Fe precipitates, along with jarosite. An unusual Pb precipitate, plumbgummite ($\text{Al}_3\text{Pb}(\text{OH})_5(\text{PO}_4)_2$) was also listed as a possible precipitate, but a discussion of this mineral was not readily obtained from the literature.

In addition to the above samples, four samples for which a pH sweep was performed were also modeled with a fixed pH. The results from these runs indicate that many of the solids predicted to form during the pH sweep runs were present at a similar equilibrium pH value as indicated by the saturation indices (Appendix E-3). The results of the pH sweeps are discussed in detail in the following subsection.

pH Sweep - No Limestone added; Fixed $p\text{CO}_2$: A sweep of pH was performed on 3 samples collected from the 40 cm depth from the control plots at sites 07, 21 and 33. Samples from site 21 and 33 were from lysimeters installed in tailings. The lysimeter at site 07 was installed in soil/alluvium underlying tailings. The sample for site 33 was 1992 pore water data (33c40). Data from 1990 was used for sites 07 (07c40) and 21 (21c40) because 1992 data was unavailable. The partial pressure of CO_2 was fixed at 0.003 atm for all runs. The initial starting point for the sweeps was the nearest one half pH unit which was lower than the measured pH value (eg: measured value = 6.87, starting sweep pH = 6.5) and the final pH was 9.0. For sample 33c40, pH was varied from 3 to 9 in one unit increments. In addition, a 1992 sample from the 90 cm depth from the control plot at site 33 (33c90) was modeled in a similar manner. This sample was collected from a lysimeter installed in natural soil beneath the tailings.

Tables 5.3.2 through 5.3.5 show the solid phase precipitation-dissolution reactions for each model run at each pH sweep value. This information is useful in discussing the control of various solid phases on metal solubility as pH varies and should be referred to during the following discussion of metal behavior.

Table 5.3.2. Solid phase reactions for pore water sample 9007c40 model run with fixed $p\text{CO}_2$.

pH	PRECIPITATION	DISSOLUTION
5.5	$\text{Fe}(\text{OH})_{2.7}\text{Cl}_3$ chloropyromorphite - $(\text{Pb}_5(\text{PO}_4)_3\text{Cl}_{(c)})$ MnHPO_4	no dissolution
6.0	cupric ferrite ($\alpha\text{-CuFe}_2\text{O}_4$)	$\text{Fe}(\text{OH})_{2.7}\text{Cl}_3$
6.5	boehmite ($\text{Al}(\text{OH})_3$)	no dissolution
7.0	tenorite (CuO)	no dissolution
7.5	octavite (CdCO_3)	no dissolution
8.0	rhodochrosite (MnCO_3) calcite (CaCO_3) $\text{ZnCO}_3 \cdot \text{H}_2\text{O}$	chloropyromorphite
8.5	dolomite ($\text{MgCa}(\text{CO}_3)_2$)	$\text{ZnCO}_3 \cdot \text{H}_2\text{O}$ MnHPO_4
9.0	no precipitates	tenorite boehmite

Table 5.3.3. Solid phase reactions for pore water sample 9021c40 model run with fixed $p\text{CO}_2$.

pH	PRECIPITATION	DISSOLUTION
6.5	tenorite MnHPO_4 gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) rhodochrosite octavite fluorite (CaF_2) boehmite cupric ferrite	no dissolution
7.0	$\text{ZnCO}_3 \cdot \text{H}_2\text{O}$ chloropyromorphite	no dissolution
7.5	FCO_3 apatite - $(\text{Ca}_{9.5}\text{Na}_4\text{Mg}_1(\text{PO}_4)_{4.8}(\text{CO}_3)_{1.2}\text{F}_{2.5})$ dolomite	MnHPO_4 gypsum chloropyromorphite
8.0	chloropyromorphite	fluorite
8.5	magnesite (MgCO_3)	FCO_3 apatite
9.0	no precipitation	chloropyromorphite boehmite

Table 5.3.4. Solid phase reactions for pore water sample 9233c40 model run with fixed $p\text{CO}_2$.

pH	PRECIPITATION	DISSOLUTION
3.0	$\text{Fe}(\text{OH})_{2.7}\text{Cl}_{.3}$ gypsum	no dissolution
4.0	alunite $(\text{KAl}_3(\text{SO}_4)_2(\text{OH})_6)$ MnHPO_4	no dissolution
5.0	no precipitation	no dissolution
6.0	bronchantite - $(\text{Cu}_4(\text{OH})_6\text{SO}_4)$ fluorite cupric ferrite	$\text{Fe}(\text{OH})_{2.7}\text{Cl}_{.3}$
7.0	octavite rhodochrosite $\text{ZnCO}_3 \cdot \text{H}_2\text{O}$ boehmite tenorite	alunite bronchantite
8.0	FeCO_3 apatite dolomite calcite	gypsum MnHPO_4
9.0	no precipitation	fluorite FeCO_3 apatite

Table 5.3.5. Solid phase reactions for pore water sample 9233c90 model run with fixed pCO₂.

pH	PRECIPITATION	DISSOLUTION
6.5	Fe(OH) _{2,7} Cl ₃ FCO ₃ apatite fluorite boehmite cupric ferrite	no dissolution
7.0	no precipitation	no dissolution
7.5	calcite	no dissolution
8.0	dolomite octavite	fluorite
8.5	no precipitation	no dissolution
9.0	rhodochrosite	boehmite

Results of the pH sweep modeling are shown in Figures 5.3.1 through 5.3.7. The solid lines are the total dissolved concentrations of metal for each site predicted by MINTEQA2 at various fixed pH levels. The open boxes filled with an X are the MINTEQA2 input total dissolved pore water concentrations for the metal at a given site. The open boxes (without an X) represent all available 1990 through 1992 total dissolved pore water metal concentration data for all sites and all depths.

Several general trends are noticeable on the graphs depicted in Figures 5.3.1 through 5.3.7. Because of the variability in both input and all pore water data concentrations between sites for a given metal, it is difficult to distinguish any clear trends in response to increasing pH (between pH 3 and 5). At low pH values, many predicted metal concentrations plot as horizontal lines. This is because MINTEQA2 uses a mass balance approach for metal speciation. If no precipitates form, the total dissolved concentration of a metal will remain the same, although the speciation may change. Therefore, the input concentration and model concentration will be identical. This does not necessarily reflect the accuracy of the model in predicting field metal concentrations but does indicate that the samples were initially undersaturated with respect to solid phase mineral containing these metals. Predicted aluminum (Figure 5.3.5) and copper (Figure 5.3.1) concentrations, however, do show good agreement with field pore water data.

When a pH value at which precipitation occurs is reached, typically at near neutral values, metal concentrations are decreased. If the same solid phase is controlling the solubility of a given metal, the predicted concentrations are very similar between sites, as would be expected given similar ionic strengths. This point is illustrated with Mn in Figure

STARS PORE WATER MODELING

Cu solubility vs. pH

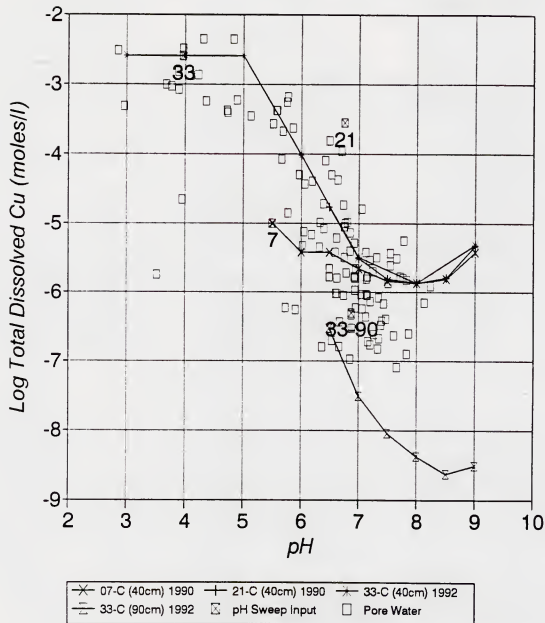


Figure 5.3.1. Total dissolved copper concentrations vs pH predicted by MINTEQA2 for several control plot pore water samples. Note that predicted concentrations are connected by line; field data shown as open boxes.

STARS PORE WATER MODELING

Zn solubility vs. pH

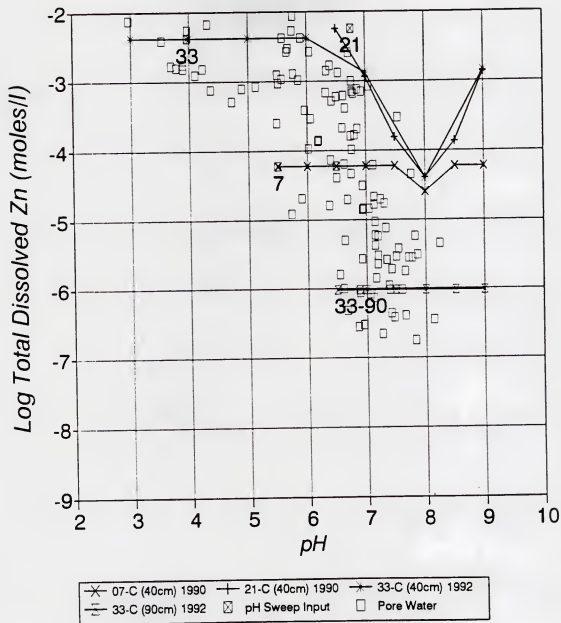


Figure 5.3.2. Total dissolved zinc concentrations vs pH predicted by MINTEQA2 for several control plot pore water samples. Note that predicted concentrations are connected by line; field data shown as open boxes.

STARS PORE WATER MODELING

Cd solubility vs. pH

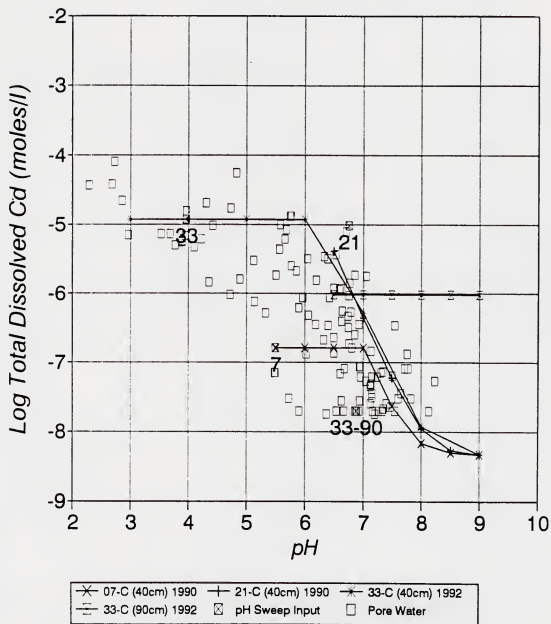


Figure 5.3.3. Total dissolved cadmium concentrations vs pH predicted by MINTEQA2 for several control plot pore water samples. Note that predicted concentrations are connected by line; field data shown as open boxes.

STARS PORE WATER MODELING Pb solubility vs. pH

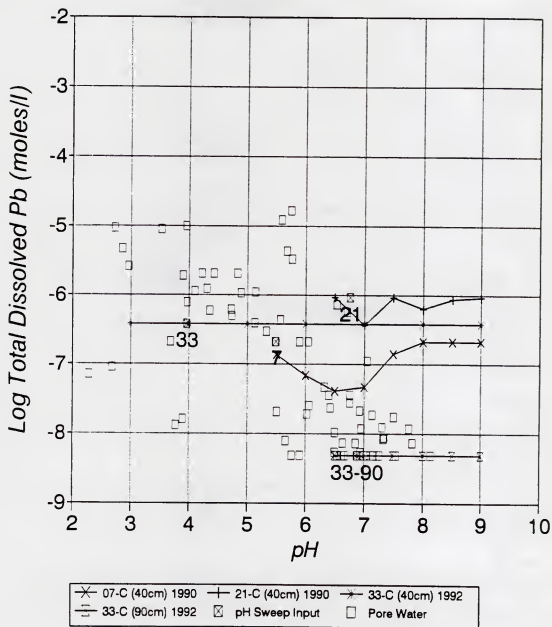


Figure 5.3.4. Total dissolved lead concentrations vs pH predicted by MINTEQA2 for several control plot pore water samples. Note that predicted concentrations are connected by line; field data shown as open boxes.

STARS PORE WATER MODELING

Al solubility vs. pH

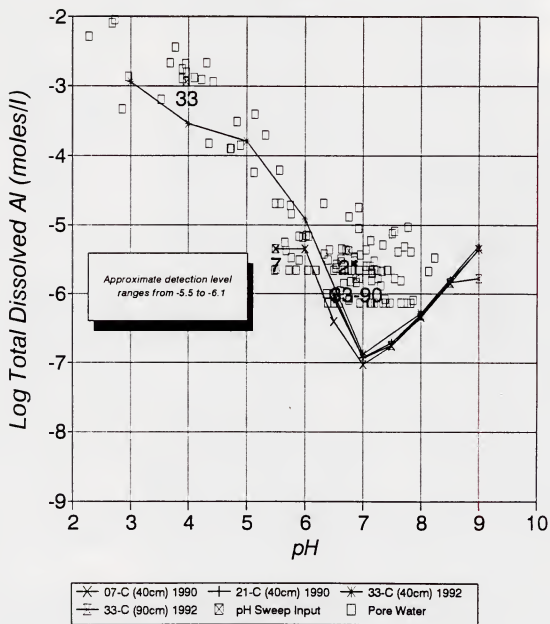


Figure 5.3.5. Total dissolved aluminum concentrations vs pH predicted by MINTEQA2 for several control plot pore water samples. Note that predicted concentrations are connected by line; field data shown as open boxes.

STARS PORE WATER MODELING

Fe solubility vs. pH

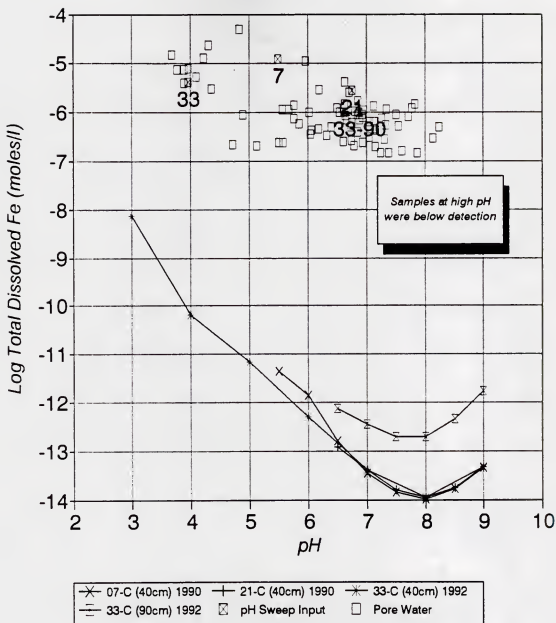


Figure 5.3.6. Total dissolved iron concentrations vs pH predicted by MINTEQA2 for several control plot pore water samples. Note that predicted concentrations are connected by line; field data shown as open boxes.

STARS PORE WATER MODELING

Mn solubility vs. pH

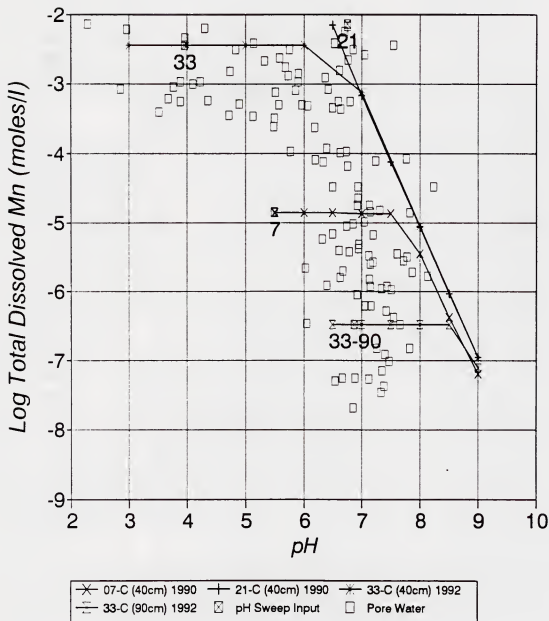


Figure 5.3.7. Total dissolved manganese concentrations vs pH predicted by MINTEQA2 for several control plot pore water samples. Note that predicted concentrations are connected by line; field data shown as open boxes.

5.3.7. Where agreement between model results and field data is strong, this suggests that the solid phase controlling solubility in the model is likely present in the field.

At pH values of 7 or greater, the model tends to overpredict the concentration of most metals relative to pore water sample data. This is because the majority of pore water samples with a pH > 7 were collected from buried soils. These soils have a metal adsorption capacity which decreases metal concentrations below those expected from a solid phase control. This is particularly true for copper and lead, and to a lesser extent cadmium and zinc. The formation of hydroxide and oxide solid phases have a stronger control in iron and aluminum solubility. Manganese is not strongly affected by adsorption reactions.

Overall, predicted metal concentrations decreased as pH levels increased, with the strongest response between approximately pH 6 and 8. This was in response to the formation of less soluble solid phases. Some metals, most notable aluminum, iron and zinc, exhibited amphoteric behavior, where soluble levels increased at high pH values after reaching minimum values at neutral pH. This implies that the addition of lime to increase pH will result in a decrease in metal concentrations, a fact readily observed in the majority of site specific field data (Appendix B3). Additionally, the neutral pH of the buried soils, in conjunction with their adsorption capacity, result in a decrease in most metal concentrations of several orders of magnitude. The soils are not, however, present at all sites (ie: not at site 7) nor can they be inferred throughout the Silver Bow Creek floodplain. Results are discussed in detail by metal below.

The speciation of soluble metals is important with regard to metal toxicity and bioavailability. In tailings samples with high SO_4 levels (eg:33c40), increasing pH resulted in a substantial change in the speciation of soluble metals (Appendix E-3). There was a decrease in free metal concentration and metal SO_4 complexes and an increase in metal hydroxy and metal carbonate complexes. The charge on the complexes varied according to the number of ligands with which the metal formed pairs. Manganese, Cu, Cd and Zn still exhibit appreciable quantities of free ion (between 32 and 61 % of total dissolved) at pH 7. Free Fe and Al are less than 3 % by pH 4. At a pH > 8 only Mn (50%) is present as a free ion. When lower SO_4 levels are present, such as in a pore water sample collected from a buried soil (33c90), a larger percentage of the metals are present as free ions or complexed with carbonate and hydroxy groups.

Specifically, Mn (61 to 67 %), Cd (42 to 50 %), Cu (61 to 64 from pH 3 to 6) and Zn (49 to 57 %) free ion comprised a significant percentage of the total soluble metal from pH 3 through 7, with the percentage changing very little with pH. The remaining soluble forms of these metals were paired almost exclusively with SO_4 . Lead was present as a free ion (20 to 37 %) or paired with SO_4 in this pH range. At pH 3, 24 % and 43 % of Al was free or paired with SO_4 , with the remainder paired with F. from pH 4 to 6, Al showed a strong affinity for F, with > 95 % of soluble Al present paired with F. Iron was paired almost exclusively with hydroxyl groups (79 and 90 % at pH 3 and 4, respectively), with > 99 of soluble Fe complexed with OH groups at pH > 5.

Figure 5.3.1 is a plot of total dissolved Cu concentration vs pH for the four pore water samples modeled. The individual points on the graph represent actual field data metal concentrations for all lysimeter pore water samples at all sites for 1990 and 1992. The low pH for sample 33c40 results in a high soluble Cu level until a pH of 6 is achieved. At this point cupric ferrite and bronchantite precipitate, considerably lowering Cu in solution. The precipitation of these same two minerals at pH 6.5 in sample 21c40 yields similar Cu concentrations, as does the precipitation of cupric ferrite in sample 07c40 at pH 6. The precipitation of tenorite, which is more soluble than cupric ferrite, at pH 7 in samples 07c40 and 33c40 does not affect Cu levels, as evidenced by the same Cu level observed in sample 21c40 as in 07c40 and 33c40, even though tenorite did not precipitate in sample 21c40. Thus, the model indicates that the presence of cupric ferrite is the dominant solid phase control on total dissolved Cu. It is worth noting the amphoteric behavior of Cu in all 40 cm samples between pH 8 and 9. This was commonly observed for most metals in this pH range and is due to the formation of negatively-charged metal-hydroxyl ion pairs with increasing OH⁻ (higher pH) (Bohn et al 1985).

In general, the model predicted total dissolved Cu levels fairly well in tailings material at pH values less than 6. At pH values greater than this, Cu levels were overpredicted by approximately an order of magnitude. This is most likely due to adsorption of copper in the underlying soil. It is interesting to note, however, that in sample 33c90, a buried soil pore water sample, predicted values are lower than measured pore water Cu levels. This sample is also the only sample in which cupric ferrite is the only copper solid phase present, which may explain the lower predicted values relative to the other samples.

The complete field data indicate that soluble Cu is undersaturated with respect to the solid phases indicated by the model. This may be due in part to Cu adsorption reactions occurring in the underlying soils. The plot of total dissolved Cu for sample 33c90 (soil underlying tailings) exhibits a very similar trend to the lines for the other samples on the graph, but at a dissolved Cu concentration approximately 2 orders of magnitude lower.

Zinc behavior was very similar in samples 21c40 and 33c40 owing to the precipitation of $\text{ZnCO}_3 \cdot \text{H}_2\text{O}$ at pH 7 (Figure 5.3.2). No change in total dissolved Zn concentration was observed in sample 07c40 until the precipitation of $\text{ZnCO}_3 \cdot \text{H}_2\text{O}$ at pH 8. At this point the soluble Zn was essentially equal in all samples. The dissolution of $\text{ZnCO}_3 \cdot \text{H}_2\text{O}$ at pH 8.5 in sample 07c40 led to a Zn concentration similar to that present before its precipitation. Samples 21c40 and 33c40 exhibited amphoteric behavior at pH > 8. Zinc concentrations did not change and no Zn solid phases formed with increasing pH in sample 33c90. This is probably the result of initial Zn concentrations which were too low to result in the precipitation of any solids.

Overall, the trend of the predicted soluble Zn lines did not follow the data points in the pH range of 3 to 7, mostly because of the variability of the field data in this range and the lack of formation of a solid phase. Above pH 7, however, the model overpredicted total dissolved zinc by between one and two orders of magnitude in the 40 cm control samples.

The majority of data points below the lines at this pH value are from buried soil pore water samples, suggesting that sorption of zinc resulted in total soluble Zn values less than those predicted by the presence of a solid phase.

Concentrations of Cd remained unchanged until the precipitation of octavite, which resulted in nearly identical total dissolved Cd in all samples at $\text{pH} > 7$ (Figure 5.3.3). Decreases in soluble Cd from approximately 10^{-5} to 10^{-8} between pH 6 and 9 occurred in samples 21c40 and 33c40. Octavite did not precipitate in sample 07c40 until pH 7, probably due to the lower relative Cd concentrations in solution. Cadmium behavior in sample 33c90 was similar to Cu in that, relative to the other samples, the concentration was approximately two orders of magnitude lower at pH 6.5 but nearly identical from pH 7.5 through 9 due to the precipitation of a solid phase (octavite in this case). The model also overpredicted total dissolved Cd by approximately two orders of magnitude relative to field Cd levels, possibly due to sorption by natural soils.

At low pH values, coinciding to tailings material at site 33, the model was in fair agreement with field data, to a pH of approximately 5. From a pH of 5 to 7 the model results were unclear relative to field data. Predicted Cd by about an order of magnitude at pH 7 underpredicted by the same amount at a pH of 8. Model results at low pH were uncertain due to site variability.

Lead concentrations showed a decrease in response to the precipitation of chloropyromorphite in samples 07c40 and 21c40 at neutral and near neutral pH values, but increased again to initial concentrations after dissolution of Chloropyromorphite at higher pH values (Figure 5.3.4). Levels of Pb in samples 33c40 and 33c90 remained unchanged, probably due to lower PO_4 concentration supported by the presence of FeCO_3 apatite (Tables 5.3.4 and 5.3.5). A total dissolved Pb concentration below detection (0.001 mg/l) for sample 33c90 would also preclude the precipitation of any solids and explains the horizontal line up of points at the bottom of the graph. Overall the model tended to overpredict total dissolved Pb by one to two orders of magnitude. As was the case for copper and zinc, this was most likely due to sorption by buried soils.

Aluminum (Figure 5.3.5), Fe (Figure 5.3.6) and Mn (Figure 5.3.7) behavior were very similar in all model runs owing to the presence of the same solid phases at each site. This was true of sample 33c90 as well. Iron concentrations were nearly identical and very low ($< 10^{-12}$) in all samples at pH values > 6 (Figure 5.3.6). This behavior was in response to the precipitation and control of Fe concentrations by cupric ferrite. The formation of the more soluble $\text{Fe}(\text{OH})_{2.7}\text{Cl}_3$ during the initial pH sweeps did not play a significant role in maintaining Fe concentrations, except in sample 33c40, where precipitation of $\text{Fe}(\text{OH})_{2.7}\text{Cl}_3$ occurred at a pH of 3. The formation of cupric ferrite at pH 6 in this sample resulted in similar Fe concentrations above this pH relative to the other samples. Although there are some ambiguities regarding the persistence of cupric ferrite in soils and the formation of $\text{Fe}(\text{OH})_{2.7}\text{Cl}_3$ relative to ferrihydrite ($\text{Fe}(\text{OH})_3$), the solubility of ferrihydrite and other common iron hydroxyoxides is such that they would support similar concentrations of Fe as cupric ferrite or $\text{Fe}(\text{OH})_{2.7}\text{Cl}_3$ (Lindsay 1979). Regardless of the solid phase controlling

Fe concentrations, the model severely underpredicted Fe concentrations in pore water relative to both the input samples and all field data. This may be due to the inclusion of colloidal Fe in the analytical sample or, more likely, to the presence of amorphous ferric hydroxide, which would support a concentration of Fe^{3+} considerably higher than predicted (Lindsay 1979).

The model input consisted of Fe^{3+} . However, the concentrations of field pore water are in the range of those supported by siderite (FeCO_3), which is a ferrous (Fe^{2+}) iron mineral. It was thought that this may explain in part the discrepancy between field and predicted total dissolved Fe levels. However, at the $\text{pe} + \text{pH}$ (16 -18) of the tailings/soil system, siderite is not stable. This is supported by the modeling performed on a saturated paste with the addition of CaCO_3 . In this model (discussed in detail in the following subsection), both Fe^{2+} and Fe^{3+} were present as components, but siderite did not form. However, the presence of maghemite (Fe_2O_3) at $\text{pH} < 6.7$ supported total dissolved Fe concentrations similar to those observed in pore water samples.

Aluminum concentrations decreased from approximately 10^{-5} at pH 6 to a low of 10^{-7} at pH 7 in the presence of boehmite (Figure 5.3.5). Somewhat higher concentrations in sample 33c40 at lower pH values were maintained by alunite, which dissolved at pH 7 in favor of boehmite. Aluminum concentrations increased at $\text{pH} > 7$ due to the presence of the hydroxy solid phase (boehmite) and increasing concentrations of OH^- (amphoterism). Overall, the predicted values of Al were close to pore water field samples, especially at $\text{pH} < 6$, suggesting that alunite is a dominant control on dissolved Al in the field at low pH values. The underpredicted values of Al at pH 7 may be due to the persistence of the more soluble alunite in the field with boehmite.

The precipitation of MnHPO_4 during initial pH sweeps maintained total dissolved concentrations of Mn at relatively constant values until the precipitation of rhodochrosite in samples 07c40 and 33c40, at which point Mn concentrations decreased (Figure 5.3.7). The wide scatter of field data makes and different input concentrations for each site makes it difficult to compare field and predicted Mn concentrations at lower pH values.

Incremental CaCO_3 Addition: A saturated paste extract sample from Ramsay Flats was used to model the effect of limestone addition on metal behavior. Limestone (as CaCO_3) was added incrementally (as aqueous Ca and CO_3), pCO_2 was fixed at 0.003 atm, Eh at 400 mv and pH was calculated by the model. Several solids were excluded including lepidocrocite, cuprite, cupric ferrite, cuprous ferrite, hematite, magnetite, goethite, Al_2O_3 , gibbsite, $\text{Fe}(\text{OH})_2 \cdot \text{Cl}_3$, diaspor, $\text{Al}(\text{OH})_3$, Zn metal and Cu metal. Solids were precluded base upon kinetic considerations. Additionally, because the model apparently was not robust enough, it was necessary to exclude precipitation of some solids to achieve convergence in some runs. Other solids were allowed to precipitate. Output analyte concentrations from a previous run were used as input for successive runs with the addition of more CaCO_3 . Input data are shown in Table 5.3.6. Output for total dissolved metal and major anion

concentrations for all model runs are presented in Table 5.3.7. Due to its large volume, the complete output is not included in Appendix E-3.

Table 5.3.6. Input data for MINTEQA2 modeling of CaCO_3 addition to 1990, 0 - 8" control saturated paste extract sample from Ramsay Flats (Site 21).

MAJOR CATIONS/pH/Eh (mv) (mg/l)		ANIONS (mg/l)		METALS (mg/l)	
Ca	292	SO_4	8800	Al	34.0
Mg	420	Cl (est)	1.0	As	0.123
Na	299	$\text{NO}_3\text{-N}$	1.1	B	0.262
K^2	2.4	F (est)	0.01	Cd	4.12
pH	3.70	$\text{PO}_4\text{-P}^2$	0.5	Cu	719
Eh	400			Fe	1.79
				Mn	1850
				Pb	0.026
				Se^2	1.9
				Zn	1337

¹ - Estimated redox potential corresponds to well-drained soil system.

² - Estimate soluble quantity based on 50% of the extractable Se concentration, or 10% of the extractable K and P concentration. Extractable Se determined using hot water extract; extractable P by the Bray method; and extractable K in 1N sodium acetate at pH 7.0. Extractable K concentration was maintained at a fixed level even if precipitation occurred due to the presence of mica and K-feldspar.

Figure 5.3.8 is a plot of pH vs cumulative CaCO_3 added and shows distinct responses of pH to added CaCO_3 . A rapid rise from pH 3.7 to approximately 6.7 occurs with the addition of only 100 mg/l CaCO_3 . In this pH range jarosite and alunite were controlling Fe and Al solubility, respectively and MnHPO_4 was present. At a pH of 4.7 antlerite ($\text{Cu}_3(\text{OH})_4\text{SO}_4$) formed, resulting in a decline in Cu concentrations (Figure 5.3.9 and 5.3.10). Gypsum was also present. The pH increased at a slower but still rapid rate between approximately 4.7 and 6.7 with the addition of a total of 800 mg/l CaCO_3 . It was within this pH range that many of the solid phases controlling metal solubility at neutral pH values precipitated. The sharp decline in Cd, Cu and Zn concentrations can be attributed to the formation of octavite, tenorite and Zn hydroxy-sulfate ($\text{Zn}_4(\text{OH})_6\text{SO}_4$) (Figures 5.3.9 and 5.3.10). Similarly, the decline in Mn concentrations can be attributed to the formation of rhodochrosite. The effect of the formation of maghemite

Table 5.3.7. Log molar dissolved metal concentrations for MINTEQA2 modeling with incremental CaCO_3 addition.

CaCO_3 (mg/l)	pH	Al+3 (mol/l)	H3AsO4 (mol/l)	Cd+2 (mol/l)	Cu+1 (mol/l)	Cu+2 (mol/l)	Cu (mol/l)	Fe+2 (mol/l)	Fe+3 (mol/l)	Fe (mol/l)	Mn+2 (mol/l)	Pb+2 (mol/l)	Zn+2 (mol/l)	SO4-2 (mol/l)	CO3-2 (mol/l)
0	3.7	-3.33E+00	-5.78E+00	-4.43E+00	-6.63E+00	-1.94E+00	-1.94E+00	-4.49E+00	-8.89E+00	-4.49E+00	-1.47E+00	-6.21E+00	-1.68E+00	-1.04E+00	-5.02E+00
20	4	-3.99E+00	-5.78E+00	-4.43E+00	-6.63E+00	-1.94E+00	-1.94E+00	-4.49E+00	-8.84E+00	-4.49E+00	-1.47E+00	-6.21E+00	-1.68E+00	-1.05E+00	-5.02E+00
40	4.22	-4.47E+00	-5.77E+00	-4.42E+00	-6.62E+00	-1.93E+00	-1.93E+00	-4.47E+00	-8.36E+00	-4.47E+00	-1.46E+00	-6.20E+00	-1.67E+00	-1.04E+00	-5.01E+00
60	4.6	-5.21E+00	-5.77E+00	-4.42E+00	-6.65E+00	-1.96E+00	-1.96E+00	-4.47E+00	-7.74E+00	-4.47E+00	-1.46E+00	-6.20E+00	-1.67E+00	-1.05E+00	-5.00E+00
80	4.64	-5.29E+00	-5.76E+00	-4.41E+00	-6.69E+00	-2.01E+00	-2.01E+00	-4.52E+00	-7.71E+00	-4.52E+00	-1.45E+00	-6.19E+00	-1.66E+00	-1.06E+00	-5.00E+00
100	4.69	-5.40E+00	-5.76E+00	-4.41E+00	-6.77E+00	-2.09E+00	-2.09E+00	-4.63E+00	-7.72E+00	-4.63E+00	-1.44E+00	-6.19E+00	-1.66E+00	-1.07E+00	-4.99E+00
200	4.84	-5.67E+00	-5.75E+00	-4.39E+00	-6.96E+00	-2.28E+00	-2.28E+00	-4.92E+00	-7.74E+00	-4.92E+00	-1.43E+00	-6.17E+00	-1.65E+00	-1.07E+00	-4.99E+00
300	4.94	-5.86E+00	-5.74E+00	-4.39E+00	-7.10E+00	-2.43E+00	-2.43E+00	-5.13E+00	-7.75E+00	-5.13E+00	-1.43E+00	-6.17E+00	-1.64E+00	-1.07E+00	-4.98E+00
500	5.47	-6.79E+00	-5.74E+00	-4.39E+00	-7.80E+00	-3.13E+00	-3.13E+00	-6.21E+00	-7.80E+00	-6.20E+00	-1.43E+00	-6.17E+00	-1.64E+00	-1.09E+00	-4.91E+00
650	6.61	-7.17E+00	-5.73E+00	-4.38E+00	-9.50E+00	-4.80E+00	-4.80E+00	-9.82E+00	-9.13E+00	-9.05E+00	-1.41E+00	-6.16E+00	-1.63E+00	-1.09E+00	-4.30E+00
800	6.72	-7.15E+00	-5.73E+00	-4.38E+00	-9.72E+00	-5.01E+00	-5.01E+00	-1.03E+01	-9.35E+00	-9.30E+00	-1.42E+00	-6.16E+00	-1.65E+00	-1.10E+00	-4.21E+00
1000	6.72	-7.15E+00	-5.72E+00	-4.60E+00	-9.73E+00	-5.02E+00	-5.02E+00	-1.03E+01	-9.36E+00	-9.32E+00	-1.44E+00	-6.15E+00	-1.66E+00	-1.11E+00	-4.20E+00
1300	6.73	-7.14E+00	-5.72E+00	-4.63E+00	-9.76E+00	-5.05E+00	-5.05E+00	-1.03E+01	-9.39E+00	-9.34E+00	-1.47E+00	-6.15E+00	-1.68E+00	-1.13E+00	-4.20E+00
1700	6.76	-7.14E+00	-5.71E+00	-4.67E+00	-9.79E+00	-5.08E+00	-5.08E+00	-1.04E+01	-9.43E+00	-9.38E+00	-1.51E+00	-6.14E+00	-1.72E+00	-1.15E+00	-4.19E+00
2700	6.79	-7.13E+00	-5.71E+00	-4.80E+00	-9.88E+00	-5.16E+00	-5.16E+00	-1.06E+01	-9.53E+00	-9.50E+00	-1.62E+00	-6.13E+00	-1.81E+00	-1.23E+00	-4.17E+00
4000	6.87	-7.11E+00	-5.71E+00	-4.98E+00	-1.00E+01	-5.32E+00	-5.32E+00	-1.09E+01	-9.68E+00	-9.65E+00	-1.78E+00	-6.13E+00	-1.93E+00	-1.31E+00	-4.14E+00
5500	7	-7.05E+00	-5.70E+00	-5.35E+00	-1.03E+01	-5.57E+00	-5.57E+00	-1.16E+01	-1.00E+01	-9.99E+00	-2.13E+00	-6.12E+00	-2.20E+00	-1.46E+00	-4.05E+00
6500	7.09	-7.00E+00	-5.70E+00	-5.60E+00	-1.05E+01	-5.71E+00	-5.71E+00	-1.21E+01	-1.02E+01	-1.02E+01	-2.36E+00	-6.12E+00	-2.44E+00	-1.59E+00	-3.99E+00
7500	7.14	-6.97E+00	-5.70E+00	-5.75E+00	-1.06E+01	-5.77E+00	-5.77E+00	-1.23E+01	-1.03E+01	-1.03E+01	-2.52E+00	-6.12E+00	-2.58E+00	-1.73E+00	-3.95E+00
8500	7.71	-6.55E+00	-5.78E+00	-6.93E+00	-6.02E+00		-6.02E+00	-1.13E+01		-1.13E+01	-4.21E+00	-6.70E+00	-3.74E+00	-1.87E+00	

SITE 21 - RAMSAY FLATS
Saturated Paste Extract (Control 0-8")

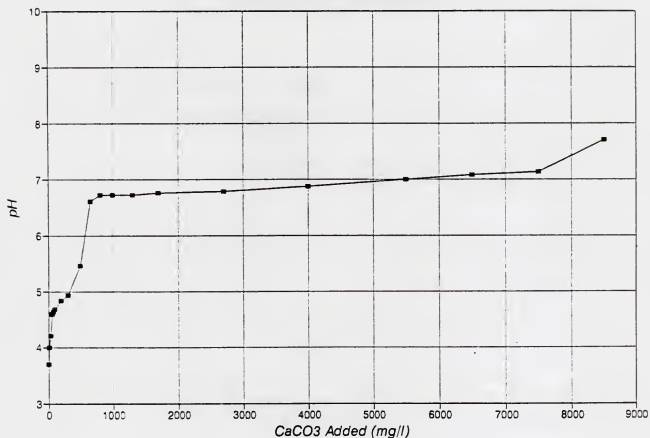


Figure 5.3.8. Change in pH with CaCO₃ addition for MINTEQA2 modeling of saturated paste extract from 0 to 8 inch depth on control plot at Ramsay Flats.

Site 21 - RAMSAY FLATS **Saturated Paste Extract (Control 0-8")**

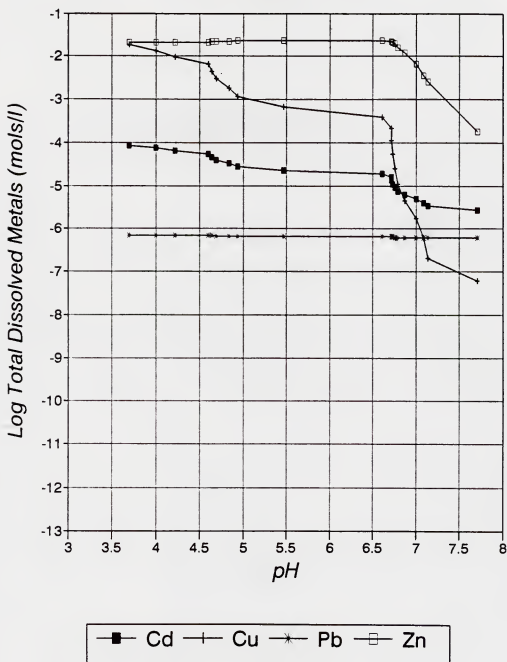


Figure 5.3.9. Predicted total dissolved cadmium, copper, lead and zinc concentrations vs pH for MINTEQA2 modeling with CaCO_3 addition to saturated paste extract from 0 to 8 inch depth on control plot at Ramsay Flats.

Site 21 - RAMSAY FLATS **Saturated Paste Extract (Control 0-8")**

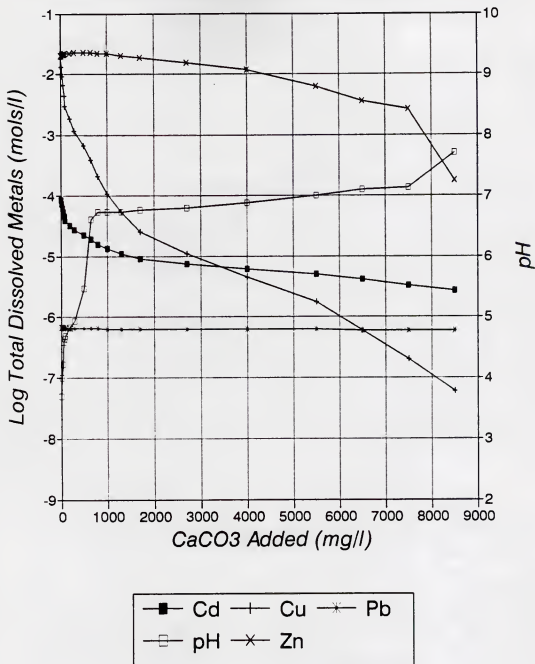


Figure 5.3.10. Predicted total dissolved cadmium, copper, lead and zinc concentrations and pH vs CaCO_3 added for MINTEQA2 modeling of saturated paste extract from 0 to 8 inch depth on control plot at Ramsay Flats.

Site 21 - RAMSAY FLATS **Saturated Paste Extract (Control 0-8")**

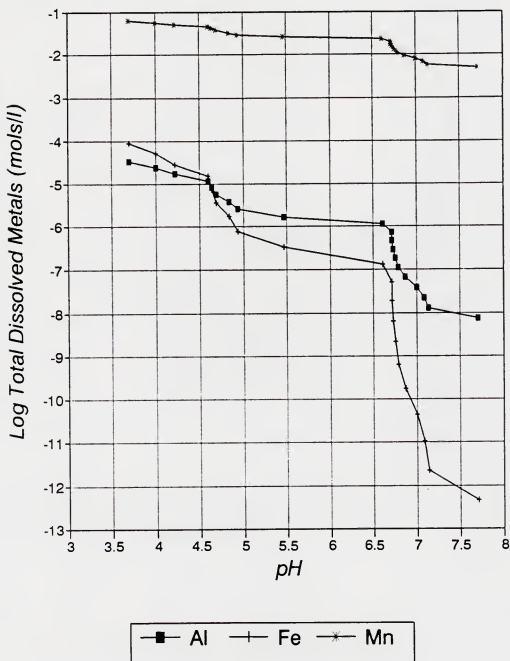


Figure 5.3.11. Predicted total dissolved aluminum, iron and manganese concentrations vs pH for MINTEQA2 modeling with CaCO_3 addition to saturated paste extract from 0 to 8 inch depth on control plot at Ramsay Flats.

Site 21 - RAMSAY FLATS **Saturated Paste Extract (Control 0-8")**

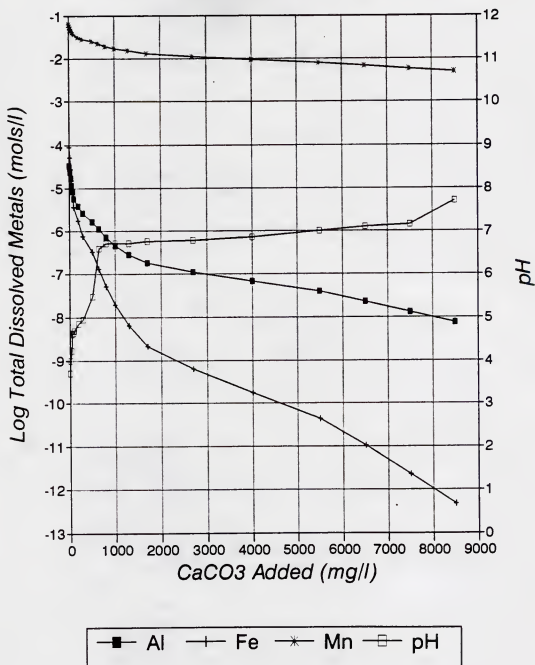


Figure 5.3.12. Predicted total dissolved aluminum, iron and manganese concentrations and pH vs CaCO₃ added for MINTEQA2 modeling of saturated paste extract from 0 to 8 inch depth on control plot at Ramsay Flats.

and boehmite at pH 5.5 on Fe and Al concentrations, respectively, can be seen in Figures 5.3.11 and 5.3.12.

The formation of these solids, particularly the hydrolysis species boehmite and $\text{Zn}_4(\text{OH})_6\text{SO}_4$, results in the production of acidity. Similarly, the formation of carbonates decreases the soluble alkalinity. The result is that the solution is buffered at a pH of approximately 6.7. This is evident in the flatter slope of the line from pH 6.7 to 8 in Figure 5.3.8, where it can be seen that a large amount of CaCO_3 addition is required to raise the pH a small amount. In fact, an addition of almost 7000 mg/l (from 1000 to 8000 mg/l CaCO_3) was required to increase the pH from 6.7 to approximately 7.5. This is due to the continued formation of solids, especially boehmite and maghemite, as can be seen in Figure 5.3.11. At pH between 6.5 and 7.0, the soluble Al and Fe concentrations drop dramatically (2 and 5 orders of magnitude, respectively) as more solid phase continues to form, resulting in the continued production of protons. Similar trends are obvious for Cd and Cu, although to a lesser degree. It is not until the concentrations of metals are severely reduced that the pH begins to increase at a faster rate. Another factor contributing to an increase in pH with decreased metal levels is the fact that the majority of soluble metals present would be paired with hydroxyl groups, thereby reducing the acidity produced by the formation of hydrolyzed soluble metals.

As is evident in Figures 5.3.10 and 5.3.12, the concentrations of Zn and Mn do not exhibit the rapid declines noted for the other metals. It is not until a pH of approximately 7.0 that zincite (ZnO) forms, leading to a sharp decline in soluble Zn concentrations. Of importance is that it would require very large additions of lime to increase the pH to a point where significant quantities of Zn and Mn would be removed from solution via the formation of solid phases.

Arsenic Behavior: Selected 1992 pore water samples with complete cation and anion data were used as input to MINTEQA2 and GEOCHEM (Sposito and Matigod, 1979; Parker et al., 1987) to determine the distribution of As species (Table 5.3.1). GEOCHEM contains an expanded thermodynamics data base including complexation reactions for As with Ca, Fe, Mg, Al, Cu, Zn, and Na; consequently, a more complete distribution of As species is predicted using the GEOCHEM database. The activity of dissolved arsenic ((HAsO_4^{2-})) for selected pore water samples was compared to a solubility plot for solid phase ferric arsenate and calcium arsenate to evaluate possible solid phase controls on pore water As. In addition, As adsorption on an Fe oxide surface was modeled with MINTEQA2 using the thermodynamic data base for Fe oxide sorption reactions contained in MINTEQA2 and Fe oxide surface properties reported in the literature (Goldberg, 1986). The sorption of As by Fe oxide was modeled using 1) measured 1992 pore water data for all cations and anions including As, and 2) an assumed total As value over a range of Fe oxide concentrations. Surface properties of the Fe oxide sorbent were obtained from Goldberg (1986) who conducted modeling studies on the sorption of As by Fe and Al oxide surfaces. The primary characteristics of the Fe oxide surface were as follows: surface area = $60\text{m}^2\text{g}^{-1}$, Site density = 1.25×10^{-4} moles g^{-1} . Suspension densities were varied between 4 grams Fe oxide L^{-1} and

50 grams L^{-1} corresponding to the values used in Goldberg (1986) and a realistic upper limit for soils and sediments in the study area:

$$\frac{(1.3 \text{ grams soil})}{\text{cm}^3 \text{ soil}} \frac{(0.01 \text{ grams Fe oxide})}{\text{gram soil}} \frac{(1 \text{ cm}^3 \text{ soil})}{0.3 \text{ cm}^3 \text{ H}_2\text{O}} \frac{(1000 \text{ cm}^3)}{L} = \frac{43 \text{ g Fe oxide}}{L}$$

Consequently, a value of 50 grams Fe oxide L^{-1} should represent realistic Fe oxide concentrations in soils/tailings where excess surface sites available for As adsorption are expected.

Ten 1992 pore water samples representing the range of total soluble As (as As(V) measured) were selected and used as input to MINTEQA2 and GEOCHEM (Table 5.3.1). Although the distribution of metal species was generally comparable between MINTEQA2 and GEOCHEM, the distribution of As species predicted using the two models varied considerably. MINTEQA2 currently does not contain thermodynamic data for the complexation of As with metals such as Fe, Al, Cu, Zn, Ca, and Mg. The GEOCHEM data base was developed specifically for application to soil systems (Sposito and Matigod, 1979) and therefore, contains an exhaustive set of thermodynamic data applicable to natural systems.

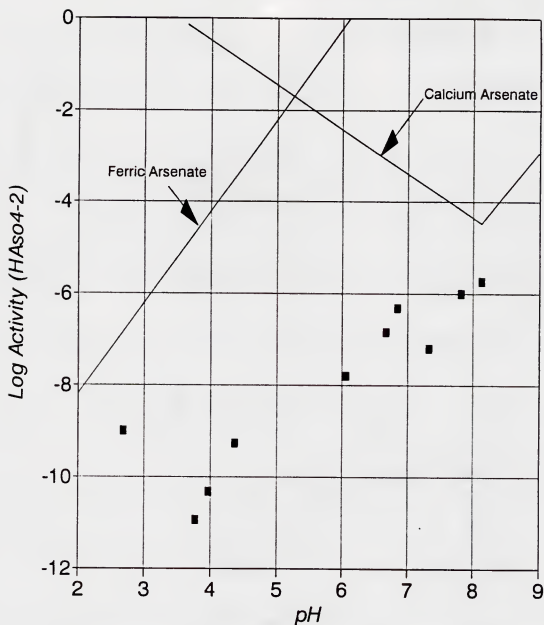
Comparisons of predicted As distribution between MINTEQA2 and GEOCHEM showed that:

1. MINTEQA2 considers only the H_3AsO_4 , $H_2AsO_4^-$, $HASeO_4^{2-}$, AsO_4^{3-} species for As(V).
2. Complexes of As(V) with Fe and Al represent an important fraction of soluble As at low pH (<5) and complexes of Mg and Ca represent an important fraction of soluble As at higher pH (e.g. >6)
3. Accurate saturation indices or predictions of solid phase control on soluble As should account for complexation of total soluble As with metal ions present in pore water samples. For example, MINTEQA2 predicted that one of the pore water samples containing high total As (7a1690) was oversaturated with respect to ferric arsenate. Conversely, consideration of metal arsenate complexes in (GEOCHEM) reduced the saturation index to well below the solubility limit for ferric arsenate.

Solid phase metal arsenates which may be important in natural systems include Ca and Fe arsenates. A solubility plot showing the predicted equilibrium $HASeO_4^{2-}$ activity (calculated using the GEOCHEM model) as a function of pH was prepared for ferric and calcium arsenate (Figure 5.3.13). Predicted activities of $HASeO_4^{2-}$ obtained using GEOCHEM for the ten 1992 pore water samples are shown on this plot for comparison and indicate that pore water samples are undersaturated with respect to those solid phases. Reduced As (III) solids were not considered since measured pe (E_h) values in pore water samples were indicative of oxidizing conditions favoring the As(V) oxidation state. Other metal arsenates

STARS PORE WATER MODELING

As Solubility vs. pH



NOTE: Data points are equilibrated pore water activities from GEOCHEM geochemical model.

Figure 5.3.13. Predicted activity of HAsO_4^{2-} in 10 representative pore water samples modeled by GEOCHEM vs solubility of ferric arsenate and calcium arsenate (from Goldberg 1986).

including Zn, Pb, Cu, and Al are too soluble under these conditions to support the activity levels of HAsO_4^{2-} observed in the 1992 pore water samples. The fact that calculated HAsO_4^{2-} activities are undersaturated with respect to these solid phases suggests that soluble As (V) concentrations are controlled by sorption reactions on solid phase surfaces, such as oxides of Fe and Al (Goldberg, 1986).

In order to evaluate the potential contribution of Fe oxide sorption of As(V) species, MINTEQA2 was used with two scenarios to model soluble As in the presence of varying Fe oxide concentrations. First, the distribution of As(V) species was predicted with MINTEQA2 using measured cation and anion data for two 1992 pore water samples over a range in Fe oxide concentrations (5×10^{-4} to 6.25×10^{-3} moles sites L^{-1} corresponding to a range in solid phase concentrations of 4 to 50 grams Fe oxide L^{-1}) (Appendix E-3). Results from the 07A90 sample (characterized by a very high total soluble As of $2.1 \times 10^{-3}\text{M}$ at a $\text{pH}=2.69$) show a linear decrease in total soluble arsenic (Table 5.3.8). At a site concentration of 6.25×10^{-3} moles L^{-1} , the predicted total soluble arsenic drops more than an order of magnitude below the value calculated in the absence of the Fe oxide surface. In a pore water sample with much lower measured total soluble As (07d150), the predicted total soluble arsenic drops from $2.2 \times 10^{-7}\text{M}$ to $6 \times 10^{-12}\text{M}$ in the absence and presence of 5×10^{-4} moles Fe oxide L^{-1} , respectively. In this case, virtually all of the total soluble As is predicted to be sorbed on the Fe oxide surface (99%) even at a low site concentration of 5×10^{-4} moles L^{-1} .

Table 5.3.8. Predicted (MINTEQA2) total soluble arsenic in equilibrium with various levels of Fe oxide. Note that sorption reactions decrease predicted soluble arsenic relative to measured soluble arsenic.

PORE WATER SAMPLE	Fe OXIDE SITE DENSITY (moles sites/l)	TOTAL SOLUBLE ARSENIC (moles/l)
Site 07,Ag Plot,90cm,1992	no sorption	2×10^{-3}
Site 07,Ag Plot,90cm,1992	5×10^{-4}	1.66×10^{-3}
Site 07,Ag Plot,90cm,1992	1.25×10^{-3}	1×10^{-3}
Site 07,Ag Plot,90cm,1992	6.25×10^{-3}	7.55×10^{-5}
Site 07,Deep Plow,150cm,1992	no sorption	2.2×10^{-7}
Site 07,Deep Plow,150cm,1992	5×10^{-4}	6×10^{-12}

One disadvantage for using this approach for evaluating the importance of As sorption in controlling soluble As is that As inputs into MINTEQA2 are based on measured total soluble As, not total soluble plus sorbed As (which is what is present under actual field conditions). Consequently, any partitioning of As into the sorbed phase will decrease the

model-predicted soluble As compared to the measured soluble As. If one could have obtained an As input value which represented both soluble As and sorbed As, then MINTEQA2 could be used to predict the partitioning of total As into soluble and sorbed phases.

An alternative approach is to *assume* a value of total As (soluble plus sorbed) for input to MINTEQA2 and predict the resultant distribution of As in solution and sorbed phases. Given a total As(V) value of $2 \times 10^{-3} \text{M}$, MINTEQA2 was used to determine total soluble As(V) as a function of pH at three Fe oxide site concentrations (Figure 5.3.14). The highest site concentration of $6.25 \times 10^{-3} \text{ moles L}^{-1}$ was estimated based on an Fe oxide content of 1%, a soil bulk density of $1.3 \text{ grams cm}^{-3}$ and a volumetric water content of 0.3. At a low site concentration of $5 \times 10^{-4} \text{ moles L}^{-1}$ the predicted total soluble As ranges from 1.6 to $1.8 \times 10^{-3} \text{M}$ over a pH range of 3 to 7. Although only 11 to 23% of the total As was sorbed by the Fe oxide, 90% of the sorption sites present were occupied by As. Consequently, the low site concentration does not represent excess Fe oxide sorption sites. Increases in Fe oxide site concentration result in greater As sorption and subsequent decreases in total soluble As (Figure 5.3.14). The sorption of As(V) species (e.g. AsO_4^{3-} , HAsO_4^{2-} , H_2AsO_4^-) at high site concentrations of $6.25 \times 10^{-3} \text{ moles L}^{-1}$ resulted in 99% of the total As being sorbed while only 20 to 30 percent of the sorption sites present were occupied by As. Consequently, this case more closely approximates what is likely occurring in the field where excess Fe oxide solid phase is available for As sorption. Furthermore, a comparison of total soluble As in 1992 pore water samples to predicted total soluble As in the presence of excess Fe oxide shows that predicted As values are within an order of magnitude of measured As values. These modeling results suggest that measured total soluble As values may be controlled by sorption reactions on Fe oxide surfaces. In addition, it is important to note that the expected increase in total soluble As (decrease in As sorption) with increasing pH (Goldberg, 1986) is more evident in the presence of excess Fe oxide ($6.25 \times 10^{-3} \text{ moles L}^{-1}$). The fact that soluble As (V) increases with increasing pH has important implications for the fate of As in amended plots where increases in pH as a result of lime addition may result in increased mobilization of As(V).

Effect of Eh on metal mobility: This section discusses the overall effects of Eh on metal solubility based on a large thermodynamic database (Lindsay 1979) and general principles of soil chemistry. As a soil/tailings system becomes more reduced the Eh decreases and the solubility of several metals in solution is affected. The metals most affected by changes in redox potential (Eh) are Fe, Mn and to a lesser degree Cu (Lindsay 1979). Reducing conditions are present when the diffusion of O_2 into soils is restricted due to the presence of water. As a frame of reference, a fairly well-oxidized soil has an Eh of about 400 mv (pe = 6.8), with a reduced system having an Eh = 0 (pe = 0). A highly reduced, waterlogged system may have an Eh of about -200 mv (pe = -3.4). Most pore water sample collected had Eh values between 200 and 800 mv and pe + pH values between 16 and 18 (Figure 5.3.15), with lower values common in 1990 samples.

STARS PORE WATER MODELING

Arsenic Adsorption by Fe Oxide

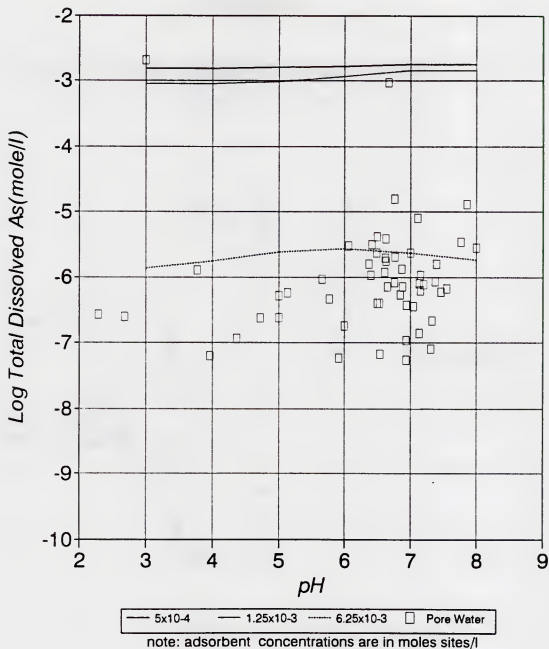


Figure 5.3.14. Predicted total dissolved concentration of arsenic using MINTEQA2 in the presence of low, medium and high iron oxide adsorbent concentrations.

STARS: SILVER BOW CREEK RI/FS **pH and pe in Pore Water (1992)**

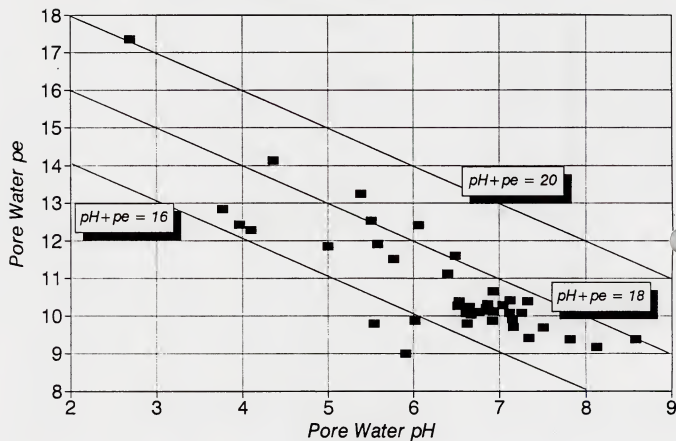


Figure 5.3.15. Plot of pe vs pH for STARS 1992 pore water data.

The term pe refers to the negative log of the electron activity ($-\log(e^-)$) and is related to Eh through the following equation:

$$Eh = 59pe \quad [1]$$

where Eh is in millivolts (mv).

In situations where the water table is close to the surface, which is common adjacent to streams and rivers, the effect of capillary rise can lead to increased water content and at least the temporary persistence of reduced conditions in tailings or underlying soils. This point was discussed in section 5.2 and was particularly applicable to site 21 Ramsay Flats. It is therefore desirable to examine the effects of reduced conditions on Fe, Mn and Cu mobility in the tailings/pore water. The following discussion is derived largely from Lindsay (1979) which contains an excellent thermodynamic database and discussion relating to the behavior of metals in soils.

Figure 5.3.16 depicts the solubility controls on Mn^{2+} of several common soil minerals as $pe + pH$ varies. The use of the term $pe + pH$ is convenient in that it provides a way of simultaneously evaluating the affect of varying these 2 parameters on metal solubility. The pH scale in the upper-right hand portion of the figure indicates that there is a two-fold log unit decrease and increase in Mn concentrations for every one unit increase and decrease in pH, respectively. As can be seen in Figure 5.3.16, rhodochrosite controls the solubility of Mn throughout most of the pH/Eh range present in the tailings/soil. Even under reduced conditions ($Eh = 0$) and neutral pH, rhodochrosite, which is not nearly as strongly affected by Eh as Mn oxides, maintains control over Mn^{2+} . Thus, a decrease in Eh in the presence of rhodochrosite is not likely to cause an increase in Mn concentrations in pore water.

What is also evident in Figure 5.3.16 is the role of pH on Mn^{2+} solubility. An increase in pH from 7 to 8 at $pCO_2 = 10^{-2.5}$ and $Eh = 400$ mv, would lead to a two-fold decrease in Mn concentrations in the presence of rhodochrosite. Similarly, at an Eh of 400 mv, the pH would have to reach approximately 8 and 10 to support the formation of manganite ($MnOOH$) and pyrolusite (MnO_2), respectively, which support lower Mn concentrations. In other words, to achieve stable, low concentrations of Mn ($< 10^{-6}$) a minimum pH of 8 would be required under oxidized conditions. As was shown in the previous section, this would require the addition of large amounts of limestone. Also, because limestone supports a pH of approximately 8.3 in equilibrium with atmospheric CO_2 , this pH represents the realistic point of Mn concentrations, although slightly higher pCO_2 levels in soils can result in an increase of up to an order of magnitude in Mn concentrations.

The solubility of Fe is also affected by Eh. At an Eh of 769 mv, the activity of $Fe^{2+}/Fe^{3+} = 1$. Thus, under the conditions present in the tailings, the majority of Fe would be present as Fe^{2+} . Figure 5.3.17 shows the activity of Fe^{2+} in the presence of ferric (Fe^{3+}) oxides, mixed ferric/ferrous oxides and common ferrous Fe minerals found in soils. As is evident, in the presence of Fe oxides and a $pe + pH$ equal to approximately 14 ($Eh = 400$ mv, $pH = 7$), Fe^{2+} (and total Fe) concentrations are maintained at low levels (10^{-11} to 10^{-15}).

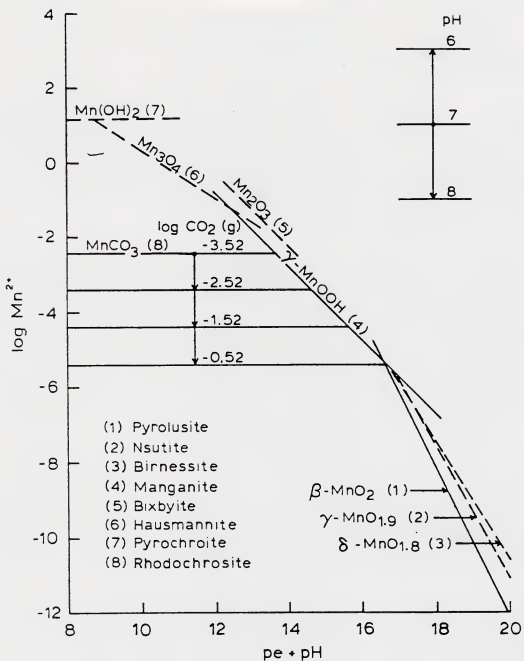


Figure 5.3.16. The effects of $pe + pH$ and $CO_2(g)$ on the solubility and stability of manganese minerals at pH 7 showing shifts of two log units for each unit change in pH (from Lindsay 1979).

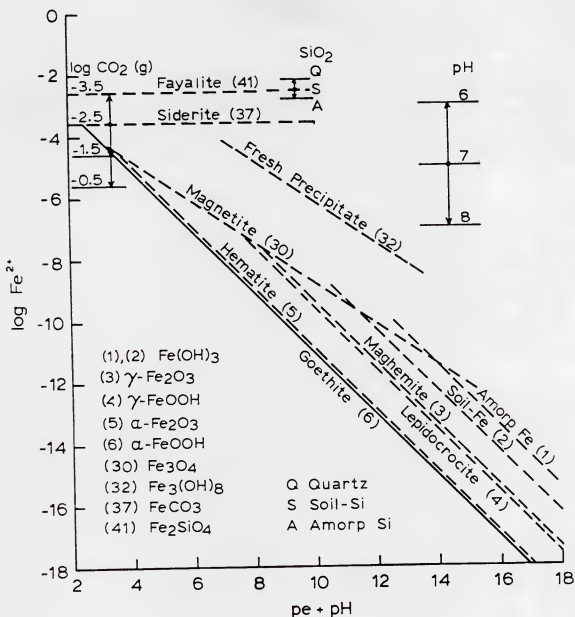


Figure 5.3.17. The effects of $\text{pe} + \text{pH}$, $\text{CO}_{2(\text{g})}$ and silica on the solubility and stability of iron minerals at pH 7 showing shifts of two log units for each unit change in pH (from Lindsay 1979).

However, under reduced conditions ($E_h = 0$), it is possible to form the more soluble Fe^{2+} carbonate mineral siderite ($FeCO_3$), which can be present at neutral pH values (until $pe + pH$ values exceed 10, or $pH = 10$ in this case). Thus under neutral pH conditions ($pH = 7$) an E_h value of 177 mv or less would be conducive to the formation of siderite. This may have implications regarding Fe mobility, especially at Ramsay Flats. However, it should be pointed out that because of the very low solubility of Fe oxides in the pH range of the soil/tailings system, transformation among Fe solid phases occurs at very slow rates.

Copper solubility is also affected somewhat by E_h . The activity of Cu^+/Cu^{2+} is equal to 1 at an E_h of 154 mv ($pe = 2.62$). Therefore, most of the total Cu in the tailings/soil pore water system was likely present as Cu^{2+} . At these levels of E_h and even under reducing conditions where Cu^+ would be the dominant form of soluble copper, the concentration of total dissolved copper would not increase appreciably (Figure 5.3.18).

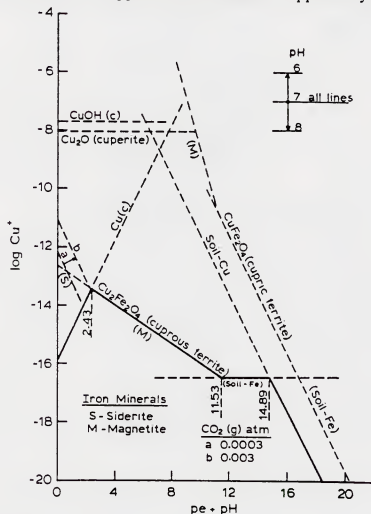


Figure 5.3.18. The effects of $pe + pH$ on the solubility and stability of copper minerals at pH 7 showing shifts for other pH values (from Lindsay 1979).

Summary: In summary, modeling efforts indicate that increasing the pH of the soil/tailings system results in a substantial decrease in metal concentrations in pore water due to the formation of solids which support lower total dissolved metal concentrations. The model also suggests that the soils underlying the tailings plays an important role in decreasing pore water metals concentrations through adsorption, as indicated by metal levels which are below those predicted if soluble metals were controlled strictly by solid phase-solute reactions. Reductions in metal concentrations are most dramatic for those sample which initially had the lowest pH and highest metal concentrations but at neutral to high pH following solid phase formation, predicted concentrations were very similar regardless of site or initial conditions. In some cases, most notably for Mn, field pore water concentrations showed large variation for a given pH value, rendering the modeling results questionable.

Metals showing the largest predicted decrease in concentration with increasing pH (pH sweep modeling runs) included Cu, Cd, Al and Fe, with decreases ranging from three to four orders of magnitude. Of the metals showing a strong decrease in concentration, copper exhibited the most rapid response to pH increase, decreasing approximately two orders of magnitude (from 10^2 to 10^4) by the time pH reached approximately 5.5. This point was illustrated by modeling results from the incremental lime addition, where a buffering plateau was observed at a pH of approximately 6.7 as the result of the formation of metal solid phases. Aluminum and iron exhibited similar behavior, although aluminum levels did not decrease substantially below pH 7. Manganese and Zn showed decreases of approximately two to four orders of magnitude, but not until the formation of solid phases at pH values greater than 7. Thus, because of the buffering effect at neutral pH values due to the formation of various solid phases, it may require substantial additions of lime to increase the pH to the point where the pore water concentrations of these metals could be significantly reduced further. Predicted concentrations of Pb decreased very little (less than an order of magnitude) relative to input concentrations and generally did not agree with field data, which showed a decrease from approximately 10^{-6} to 10^{-8} from low to neutral pH.

In tailings samples with high SO_4 levels, increasing pH results in a substantial change in the speciation of soluble metals. There is a decrease in free metal concentration and metal SO_4 complexes and an increase in metal hydroxy and metal carbonate complexes. Manganese, Cu, Cd and Zn still exhibit appreciable quantities of free ion (between 32 and 61 % of total dissolved) at pH 7. Free Fe and Al are less than 3 % by pH 4. At a pH > 8 only Mn (50%) is present as a free ion. When lower SO_4 levels are present, such as in a pore water sample collected from a buried soil, a larger percentage of the metals are present as free ions or complexed with carbonate and hydroxy groups.

Increasing the tailings pH through lime addition in conjunction with the adsorptive capacity of buried soils leads to a significant reduction in metals in pore water at neutral pH values. This suggests that the in-situ treatment investigated in the STARS program would be effective in decreasing the potential for metals flux to groundwater. The exception to this trend is arsenic, which is controlled primarily by adsorption reactions with metal oxides. Arsenic concentrations in the presence of an Fe oxide adsorbent were relatively unaffected by an increase in pH.



6.0 SUMMARY AND CONCLUSIONS

The three years of monitoring data presented in this report were collected primarily to evaluate the effectiveness of STARS treatments in eliminating or reducing the movement of metals from tailings to the surrounding environment. Both short-term and long-term effectiveness were investigated. The short-term effectiveness of the treatments was gauged through the direct measurement of soil and water physical and chemical parameters and certain vegetation parameters. The long-term effectiveness was extrapolated using standard modeling techniques on three primary environmental parameters: surface water runoff, unsaturated flow through the vadose zone, and soil geochemistry.

The STARS monitoring program was designed to compare soil amendment and tillage treatments against untreated tailings conditions and to measure relative differences among treatments. The investigation was conducted at five locations representing the range of tailings conditions found along Silver Bow Creek within the Streamside Tailings Operable Unit. Due to the extreme heterogeneity encountered at the study sites, only general conclusions can be made with respect to certain environmental media. However, some statistically significant differences were evident and many trends could be confirmed through detailed evaluation of the data.

This section summarizes the conclusions drawn from the analysis of monitoring data collected during the three year monitoring period. It also discusses the general conclusions that are supported by both the monitored data and the environmental modeling used to investigate long-term trends. Section 6.1 summarizes the conclusions drawn from the basic investigative elements of the study. Section 6.2 discusses trends evident from the detailed modeling of the specific environmental media, focusing on issues related to long-term effectiveness of the treatments. Section 6.3 combines the investigative data and the modeling to develop general conclusions regarding the overall effectiveness of STARS technologies to address the contaminant transport pathways identified in the conceptual model.

6.1 INVESTIGATIVE ELEMENTS

Investigative elements consist of actual parameters measured. These key elements include soil hydrological monitoring, soil chemistry, and vegetation success.

6.1.1 Soil Hydrological Monitoring

The soil hydrologic monitoring was conducted to develop a water budget for each STARS treatment and control tested. The water budget quantifies hydrologic parameters

at the sites, such as infiltration, runoff, leachate generation, and plant water use. The water budget is used to determine whether there is net downward movement of precipitation to the groundwater or net upward movement of groundwater driven by evapotranspiration.

Soil Water Content: Soil water content was measured at each site primarily to support the development of the water budget for each STARS treatment. Measurements were taken biweekly to bimonthly at each monitored plot by use of a neutron probe.

Unvegetated tailings at all of the sites had measured water content at or above field capacity. The capillary fringe above the groundwater was detected at all streambank tailings sites except Fairmont which is in an upland setting. The high water contents which do not vary significantly from season to season, indicate that bare tailings have little capacity to store infiltrating rainfall or snowmelt, hence increasing the likelihood of groundwater recharge. Water balance results, although somewhat inexact, suggest that 1.5 to 3 inches of recharge occur annually in bare tailings. Neutron probe monitoring results, as well as physical differences between the sites, indicate that coarse tailings material represented by the Rocker site have the greatest potential for recharge while finer tailings, as typified by the Ramsay site, have less potential. Evaporation from the bare soil surface only dried the soil profile of untreated tailings to several inches in depth. This finding is consistent with most references (Hillel 1980). In addition, unsaturated flow modeling of streambank tailings (Section 5.2) also suggest that approximately 3 inches of groundwater recharge would occur annually in unamended tailings.

Establishment of vegetation appeared to decrease the potential for recharge to occur. On treated, vegetated tailings, soil moisture may be removed by vegetation to depths as great as 3.5 feet even though the maximum vegetative rooting depth measured at the end of the three-year monitoring period was 15 inches. The magnitude in the reduction in recharge may be directly correlated with the productivity and vigor of the established vegetation. All control plots had low or zero biomass production as well as small soil water deficits at the end of the 1991 growing season. Measured soil water deficit of approximately three inches corresponded to plots with production of roughly 2200 to 2400 kg/ha of aboveground biomass. This is potentially enough deficit to offset or prevent groundwater recharge. The increased plant water use on some plots may have been partially offset by an increase in infiltration rate, especially in the deep-till treatment at Opportunity.

Rainfall Simulation: Rainfall simulation tests were conducted at three of the STARS sites: Rocker, Ramsay, and Opportunity. Surface runoff water volume and quality were measured during these tests. Rainfall simulation tests completed for this study used a steady state rate of 1.5 to 2 inches of rain per hour which is equivalent to the 100 year, 6 hour event for all tested sites and in excess of the 100 year, 24 hour event for all tests except the agricultural tillage plot at Ramsay. Rainfall simulation results showed that all the treatments implemented at Ramsay and Opportunity have some ability to both reduce runoff volumes and improve runoff water quality (when runoff occurred) compared to the control

treatment. Treatment effects were not apparent at Rocker where coarse textured soils promoted infiltration and runoff did not occur from any of the plots including the control.

The cover soil, deep tillage, and agricultural tillage treatments show the highest potential to reduce runoff and improve the quality of runoff. The cover soil treatment would appear to be the most effective of those tested in that it eliminated runoff under the rainfall application rate tested. The cover soil treatment essentially eliminated runoff at Ramsay. The deep plow treatment was nearly as effective in that only one of three sites tested, Ramsay, produced runoff. Compared to deep tillage, agricultural tillage produced slightly lower runoff with poorer quality at Ramsay and slightly higher runoff with poorer quality at Opportunity. The injection treatment actually resulted in increased runoff when compared to the control at Rocker. However, the increase in the quality of runoff on the injection treatment still produced a reduction in metals contained in runoff when compared to the control.

Specific site differences were readily evident. At Rocker, most applied rainfall on both treated plots and the control plot apparently infiltrated into the coarse textured soils. The calculated curve number for the control plot at this site was very low (6.2). This suggests that most precipitation falling at Rocker, if not utilized by evapotranspiration demands, will percolate to the groundwater system. At Ramsay, the untreated tailings curve number (94) was similar to that expected for hard surfaced roads and most precipitation could be expected to contribute to runoff. All applied treatments were at least somewhat effective in reducing the runoff potential, with the ag-till and deep plow plots exhibiting curve numbers similar to native range land in good condition. The cover soil treatment essentially eliminated runoff at Ramsay. These findings suggest that a large portion of the precipitation falling on untreated tailings at Ramsay will runoff and that all STARS treatments, except the slurry injection, will markedly decrease potential runoff at the site. At Opportunity, conditions were similar to Ramsay with respect to the relative volume of runoff produced. The control plot again exhibited a curve number typical of hard surfaced roads while both the cover soil and deep plow treatments eliminated runoff at the site under the applied rainfall rates. Runoff from the ag-till and injection plots was similar to typical range conditions.

The predominant chemical form of metals and AS contained in runoff from amended plots was in the total recoverable fraction. In unamended plots, metals in runoff were predominantly in the dissolved fraction. This result indicates that soluble metal salts have probably formed on unamended tailings and are dissolved into surface runoff during precipitation events.

6.1.2 Soil Chemistry Monitoring

Soil chemical data was collected from all sites to measure the ability of the STARS treatments to neutralize tailings acidity, the bioavailability of the metals in each STARS treatment, and the relative mobility of the metals.

Pore Water Chemistry: Samples of pore water were collected for use in conjunction with the water budget information to develop an estimate of metal movement out of the tailings as leachate from each STARS treatment. Samples were collected from the unsaturated zone with suction lysimeters. Significant spatial and temporal variability characterized much of the pore water data; however, several important conclusions can be made about the geochemistry of the control and amended plots at each site. While solution pH is an important variable which affects the solubility of most metal cations, the pH of the vadose zone can vary through time by two to three units, potentially due to a number of factors. Increases in biological activity can increase dissolved CO_2 levels as the temperature warms which may lead to a decline in pH. Conversely, plant growth may dry the soil leading to an increase in the gas exchange rate causing a decline in dissolved CO_2 . Sudden and extreme decreases in pore water pH with concurrent increases in EC were thought to be due to movement of pore water from upper soil layers where ferric sulfates and other salts dissolved leading to hydrolysis of iron and a decline in pH. It is interesting to note, however that water quality data, particularly metals concentrations, did not vary substantially from year to year on most of the treatments monitored.

While the generally high spacial variability in data and the lack of replication prohibit drawing statistical comparisons among treatments, general trends in the pore water chemistry data are evident. Amended plots generally showed an increase in pore water pH with the dominant cations being calcium and magnesium and the dominant anions being sulfate and bicarbonate. Trace metal levels were shown to decrease on amended plots. The dominance of pore water chemistries by the alkaline earth elements on treated plots was due to the addition of lime material, although downward leaching of amendments (alkalinity) was negligible. Arsenic concentrations were observed to increase at depth in the amended plots at many of the monitored sites, primarily a result of increasing pH.

Concentrations of soluble aluminum, iron, and copper were substantially reduced in pore water as pH increased while, levels of manganese, cadmium, zinc, and lead did not have a clear correlation with increasing pH until pore water pH could be raised to levels greater than 7.0. This data indicates that much higher amendment rates, or different base amendments such as sodium carbonate, may be needed to substantially reduce concentrations of cadmium, manganese and zinc. Lead concentrations in pore water were generally at concentrations near the laboratory detection limit and do not appear to be a concern in the vadose zone.

In general, no difference in pore water chemistry was observed between the control,

agricultural tillage and cover-soil treatments. The deep plow treatment often exhibited an increase in solution pH and a decrease in iron, copper, and cadmium levels compared to the control plot at both the 40 and 90 cm depth. Less often, a decrease in manganese and zinc levels and an increase in arsenic levels were also noted in the deep plow treatment compared to the control. Slurry injection appeared to be less effective in controlling solution pH at most sites than any of the other treatments tested.

At the Manganese Stockpile site and at Rocker, tailings were probably lying with the saturated zone, hence little or no separation occurred between the water table and tailings material. At both of these sites, elevated levels of metals were detected in the deepest lysimeters (150 cm), well below the tailings zone. Downward leaching of metals and As had clearly caused localized groundwater contamination at these sites, at least in the surficial portion of the aquifer. Apparent contamination of sediment underlying the Rocker tailings may have been due to the coarse texture of the natural sediments which exhibit little buffering or metal ion adsorption capacity. At Ramsay and Opportunity, tailings are 50 and 30 cm thick, respectively, while groundwater lies at least 100 cm below the tailings/natural soil interface. This distance between the base of the tailings and the water table and the presence of buffering capacity in the native soils provides a geochemical barrier which attenuates metals that would otherwise leach to groundwater. Natural soils at these sites have confined metal contamination to the upper 30 to 50 cm of buried sediments.

Soluble Metals in Soil: Soil samples were collected at various depths from the treatment plots at all sites during 1990. The samples were analyzed for concentrations of water soluble metals to help evaluate the potential for each STARS treatment to reduce or inhibit the movement of metals to groundwater. Changes in soluble metals and As concentrations over the three year period indicate the relative ability of the treatments to reduce leachability of the tailings material. Soluble metal concentrations, including Cd, Cu, and Zn in the 0 to 15 cm depth interval were dramatically reduced by the STARS treatments at the Manganese Stockpile, Ramsay, and Opportunity. Statistically significant reductions in these three metals concentrations for all the treatments were measured at Ramsay when compared to the control. Water soluble lead concentrations did not differ markedly between the treatments and the control at any of the sites. Water soluble As concentrations increased significantly in the 0 to 15 cm depth interval at Ramsay in the treated plots compared to the control, probably due to the increased mobility of As at a pH near 7.0. Soluble As concentrations also increased in the 0 to 15 cm depth interval at Fairmont in the treated plots but significantly only in the extra plot treatment.

Soluble Cd, Cu, and Zn concentrations in the 15 to 60 cm depth interval were higher in some treatments and lower in other treatments than the control at each of the five sites. Significantly lower metals concentrations than the control in this depth interval were noted at the following sites and treatments: at the Manganese Stockpile for the deep plow and coversoil treatment (Cd and Cu) and the deep plow treatment (Zn); at Ramsay for the injection treatment (Cu); and, at Opportunity for the deep plow and coversoil treatments

(Cd and Zn) and all the treatments (Cu). Soluble As and Pb concentrations were not significantly lower than the control in this depth interval except for the deep plow treatment at the Manganese Stockpile and Opportunity (Pb only).

Soluble metal and As concentrations in the 60 to 120 cm depth interval were not significantly lower than the control at any of the sites. However, the deep plow treatment did have significantly higher metals concentrations than the control for all the metals except As in this depth interval at the Ramsay site. It is presumed that the deep plow caused this increase in soluble metals by mixing highly contaminated surface tailings (including metal salt crusts) with native materials in this depth interval.

Extractable Metals in Soil: Changes in extractable metals and As concentrations over the three year period indicate the relative ability of the treatments to reduce phytotoxicity of the tailing material. Extractable concentrations were measured in the same two depth intervals tested for the soluble metals. Extractable (or bioavailable) metals and As concentrations did not uniformly show the considerable differences noted for soluble metals and As concentrations between the treatments and the control. Statistical differences could not be measured due to the lack of replication. At Rocker and Ramsay, there was generally only relatively small differences between the treatments and the control. At these two sites there was also little difference between the extractable concentrations measured at the two depth intervals (0 to 15 cm and 15 to 46 cm). At Rocker, the relatively high infiltration rates in the coarse textured tailings probably leached metals from all plots sampled, thus resulting in somewhat similar results. Extractable metals concentrations in treated plots at Fairmont and Opportunity generally were lower than the control, especially for Cu, Pb, and Zn in the 0 to 15 cm depth interval. Differences in extractable metals and As concentrations with depth were generally relatively small at these two sites at the deeper interval.

While the extraction methods used are meant to simulate the slightly acid chemical conditions of the rhizosphere, the buffering procedure used in the extraction may not completely represent pH conditions found in native or tailings material, especially in the well buffered soils present at the Ramsay and Opportunity sites. However, the purpose for obtaining the extractable (bioavailable) data was to ascertain if this parameter related to metal and As concentrations in vegetation. As will be discussed further under the vegetation summary, extractable concentrations in soil generally correlated positively with metals and As concentrations measured in and on plant tissue.

Saturated Paste pH and Electrical Conductivity: Mean saturated paste pH and EC were measured in 1990 and eight depth intervals at each of the sites. These measurements were taken to assess the success of tailings neutralization achieved by the STARS treatments. Measured pH values in the 0 to 15 cm depth interval were higher than 7.0 standard units in all treatments at each of the five sites. Measured pH values in treated

plots were generally higher than a pH of 6.0 and higher than the control in the 0 to 30 cm depth interval. At most of the sites, the pH of the control and the treated plots were similar at depths greater than the 30 to 60 cm depth interval. Generally, these depths were below the amendment incorporation zone except for the deep plow treatment where amendments were mixed to a depth of approximately 100 to 120 cm.

Addition of lime amendments at all sites generally increased the EC of the amended plots compared to the control. Below the zone of incorporation, the EC of the treated plots was similar to the control. Although Ec levels were highly variable, they generally ranged between 2 and 12 dS/m. At all the sites except rocker, treated plot EC values in the 0 to 15 cm interval were generally above 4 dS/m, a level generally considered to be saline by the U.S. Department of Agriculture.

Mean saturated paste soil pH and EC measured on amended plots from the 0 to 15 cm depth interval in 1990 and 1991 indicated that there were no significant changes in these parameters during this period. The stability of soil pH and EC measurements through time contrast with the high variability of pH and EC measured in pore water through time.

Total Metals in Soil: Total metals and As concentrations were measured in the 0 to 2.5 cm surface interval during the third year of monitoring. The measurements were taken to evaluate whether differences in surface metals levels could be identified or attributed to the STARS treatments. Total metals concentrations in these surface samples generally exhibited non-significant differences between amended treatments, the control, and undisturbed tailings surrounding the study plots at all the sites except Fairmont. At the Fairmont site, the deep plow treatment significantly reduced total As, Cd, Cu, and Pb concentrations compared to agricultural tillage, the extra treatments, and the unamended control. The coversoil material had significantly lower (except non-significantly lower for Cd at Rocker and Opportunity) concentrations of total metals than any of the other treatments at all sites where it was implemented.

Total metals and As characteristics observed at Rocker, Ramsay, and Opportunity are primarily a result of tailings material being generally thicker than the depth of amendment incorporation. This situation probably limited the effectiveness of dilution through mixing. At Fairmont, metals contaminated material is limited to the upper 10 to 20 cm. Because of this, agricultural tillage (which was utilized on the control, injection, deep plow, and extra plots) generally lowered metals concentrations, although not significantly for all metals, compared to the undisturbed samples. Deep tillage at Fairmont reduced total metal and As levels by nearly an order of magnitude compared to the undisturbed condition. Surface chemistries observed in this one sampling in the spring quantify the total metals and As at a specific time and within a specific hydrologic regimen. Total metals concentrations may also be affected by the formation of soluble salts on the surface. As a consequence, total surface material metal levels may fluctuate as a function of chemical, physical, and environmental factors.

Analysis of total metal levels in treated and untreated streambank tailing areas confirmed that STARS treatments were not an effective means to reduce surface metal levels where the depth of tailings was equal to or greater than the depth of tillage. Covering the tailings with coversoil resulted in a decrease in surface metal levels.

6.1.3 Vegetation Monitoring

Vegetation became established on all amended treatments at all of the sites except the Manganese Stockpile. The combination of flooding, drought and high soil salinity at the Manganese Stockpile killed most of the established vegetation within the first year and subsequent monitoring was discontinued. Plant performance at the remaining sites was measured using areal cover and production by species and further evaluated using measurements of seedling germination and survival, plant rooting depth and plant tissue metals and As concentrations. Importance values were calculated by summing the relative cover and production by species to represent a single measure of the status of specific species on each treatment at the time of sampling.

Germination and survival was sufficient to establish vegetation on all amended treatments at the four sites carried through the three year monitoring period. Established vegetation set and dispersed seed during the first growing season. At almost every site, vegetation production and cover on the treated plots was significantly higher than production and cover measured on the control plots. There was no significant difference at any of the sites between cover and production in the 15 cm thick coversoil and the 38 cm thick coversoil. Vegetation cover and production on the deep plow treatment were usually comparable to cover and production measured on the coversoil treatment. Vegetation cover and production on the agricultural tillage treatment at some sites were not significantly different than cover and production on the deep plow treatment.

Importance values for highest seeded species at each of the four sites where vegetation characteristics were measured is summarized in Table 6.1. These species exhibited better cover and production than the other seeded species. However, long-term survival of these species can not be determined from the three years of monitoring data collected during the study nor do importance values take into account the effect of plant succession.

Table 6.1. Species exhibiting highest Importance Values.

Site	Species	Overall Rating ¹
7	Mammoth wildrye	109.2
	Altai wildrye	103.2
	Thickspike wheatgrass	36.4
21	Tall wheatgrass	184.9
	Intermediate wheatgrass	125.3
	Russian wildrye	59.5
27	Desert wheatgrass	144.9
	Crested wheatgrass	101.3
	Russian wildrye	72.8
33	Altai wildrye	103.7
	Basin wildrye	51.8

¹ Mean of all measured importance values for each species.

Metals and As concentrations in and on plant tissue varied from site to site. At Rocker, no significant differences in metals and As concentrations were detected between any of the treatments for altai wildrye and relatively few significant differences in metals and As concentrations were detected between any of the treatments and the control for thickspike wheatgrass.

Metals and As concentrations at Ramsay were generally three to ten times higher than those at Rocker. Measured Cu, Cd, Pb, and Zn concentrations on vegetation at Ramsay are high enough to potentially cause problems for grazing animals. While metals and As concentrations in the coversoil treatment were significantly lower than the other treatments at Ramsay for many of the metals (especially on intermediate wheatgrass), metals and As concentrations were substantially higher in plant tissue on the coversoil treatment at Ramsay than at Rocker. Because of this and the fact that the Ramsay site is surrounded by a large expanse of bare tailings, these higher metals concentrations are likely a result of metals and As dust and rainfall splash on the vegetation rather than in the vegetation.

Metal and As levels were generally lower in and on vegetation at Fairmont and Opportunity than at Rocker or Ramsay. However, as at Ramsay, Cd concentrations were high enough to pose a potential problem to grazing animals at both sites. Cadmium was also found to be elevated in vegetation tissue during the greenhouse trials (STARS Phase I).

To further evaluate the relationships between bioavailable metals and As concentrations in amended tailings and these levels found in vegetation, plant tissue metals

and As concentrations were correlated with soil extractable metals and As concentrations. Regression statistics were compared for each of the wheatgrass species, for the wildrye species, and for all plant species. Correlation coefficients for the wheatgrasses and all plants were positive, had similar slopes, and were significant for Cu, Cd, Mn, Pb, and Zn. Correlation coefficients for ryegrass alone were also positive and significant for these same metals but the regressions coefficients were lower and exhibited a different slope than the wheatgrasses and all plants. Significant correlations indicate that metal loadings in and on the vegetation are related to extractable levels found within the amended root zone of the tailings.

Arsenic found in ryegrass plants was positively correlated to and significant with measured bioavailable soil As levels. The amount of As found in wheatgrass samples was negatively correlated (nonsignificant) to extractable soil concentrations. Reasons for the difference in As correlation between the wheatgrass and the wildrye grasses is unknown. When the data for As were combined for all plants, the r value was -0.082 indicating no relationship between bioavailable As was identified.

After three years of monitoring vegetation, rooting depths were highly variable from treatment to treatment but, generally, the majority of the root mass was located in the upper 6 to 10 inches in the amended plots. Vegetation rooting had progressed to a maximum depth of 15 inches in the deep plow plot at Opportunity. Roots almost always limited themselves to the zone amended, following zones of high pH and avoiding zones of low pH.

Plant communities established on streambank tailings areas will likely become subirrigated hence their production is expected to remain fairly consistent from year to year as long as the chemical and soil nutrient conditions and grazing management (if initiated) are conducive to maintenance of stand. Periodic flooding of streambank tailings areas or presence of very high water tables may limit root zone depth or may otherwise prevent plant establishment (due to salinity for example) thus limiting the benefit of STARS treatment in areas such as the Manganese Stockpile. Chemical amelioration may improve the quality of surface runoff or groundwater recharge notwithstanding a general failure of the vegetative stand, however.

Amendment Mixing Summary: At the conclusion of the three-year monitoring period, test pits were excavated by backhoe in representative STARS treatment plots at the Rocker, Ramsay, and Opportunity sites. A pH indicator dye was sprayed on the face of the excavations and visual and photographic observation of soil pH conditions made. From the observations, the degree of mixing achieved by the incorporation methods, the volume of soil neutralized, the continuity of acid and neutral zones, and the plant root growth patterns could be evaluated. Of the three major types of treatment methods utilized (agricultural tillage, deep tillage, and injection), both the agricultural tillage and deep plow methods produced acceptable results in terms of amendment mixing.

Excavations of deep plow plots clearly showed a more even distribution of amendments (compared to injection plots). Amendments were incorporated to a depth of 3.5 feet, although full distribution of the amendments throughout the plot was not consistently obtained with the single-pass tillage used at the STARS plots. Multiple passes with the deep plow could improve the uniformity of distribution of amendments within the treated zone.

Agricultural tillage was successful in incorporating the amendments to a depth of approximately eight inches and uniformly distributing the amendment material in this zone. The agricultural tillage treatment may be very useful for treating shallow tailings.

The injection treatment was less successful in achieving adequate mixing. The injected lime slurry tended to follow only the injection boreholes and occasional stratifications, moving into the soil profile through the largest soil pore channels. As a result, vertical and lateral lime stringers were formed and much of the plot subsurface remained unamended. Additionally, due to back pressure when impermeable materials were encountered, some of the slurry was rejected from the injection bores and ponded on the surface.

6.2 PREDICTIVE MODELING

6.2.1 Precipitation Runoff and Erosion Modeling

The U.S. Department of Agriculture GLEAMS model was used to estimate changes in precipitation-induced runoff and erosion due to chemical treatment and revegetation of streambank tailings. The model was calibrated by measuring simulated rainfall runoff and observing site soil characteristics, hydraulic, topographic, and other factors. The water balance routine within GLEAMS did not parallel field-measured results at the STARS site. In general, predicted winter soil evaporation was much higher in GLEAMS than that measured in the field. Suggestions for improving the water balance portion of GLEAMS are contained in the text. Nonetheless, erosion and surface runoff estimates derived from the GLEAMS model were expected to provide useful comparisons between sites and between treatments.

Ramsay Flats: Runoff and erosion was simulated for the control and deep plow plots at Ramsay flats using the calibrated GLEAMS model. Significant differences in the amount of runoff and erosion were predicted between the control and deep plow treatments. For the three year simulation, 1.46 inches of runoff occurred each year from the control while only 0.67 inch was predicted from the deep plow plot. The timing of runoff also differed between the control and deep plow treatments. Peak monthly runoff for both treatments occurred in April or May depending on the year. Measurable amounts of runoff

occurred throughout the summer from the control plot while no runoff occurred after the end of May from the deep plow plot. This difference in the timing of runoff is thought to be significant in that most fish kills on the Clark Fork have been observed in July and August when convective thunderstorms caused runoff of metal salts which had accumulated on the tailings surface after prolonged dry periods.

Significant differences in erosion rates were also noted. For the control site, an average annual soil loss of 13.7 tons/acre was predicted while only 1.2 tons/acre was predicted for the deep plow site. The ten-fold reduction in erosion was due to the reduction in runoff as well as the protection provided by the vegetative cover established on the deep plow plots.

Rocker: Simulation of runoff and erosion from the control and deep plow plots at Rocker was also conducted. The soil material at Rocker is coarser in texture than at Ramsay Flats and has a lower water-holding capacity and a higher infiltration rate. Hence, the Rocker soils are much less erosive than soils at Ramsay Flats. The water balance results for three years of simulated rainfall at Rocker indicated that 1.69 inches of percolation would occur on the control plot, while the higher evaporative use of water by established vegetation in the deep plow plot would prevent percolation. Runoff averaged 0.66 inch and 0.61 inch on the control and deep plow plots respectively. Predicted runoff was less at Rocker than at Ramsay Flats due to the lower runoff curve number of both the control and deep-plow plots at Rocker.

Predicted soil loss at the Rocker site was 0.7 tons/acre for the control and 0.14 tons/acre for the deep plow plot. These low rates of soil loss were due to the coarse texture and rapid infiltration rates of soil at Rocker. The lower soil loss predicted for the deep plow plot is due to the vegetation established on the deep plow plot.

Opportunity: GLEAMS simulation of the Opportunity control site yielded and estimated annual 1.69 inches of runoff and 0.42 inch of percolation. Runoff occurred intermittently from March through September with peak contribution in April. Percolation occurred from April through June when the soil profile was at its annual maximum.

The deep plow plot at Opportunity had substantially less predicted runoff (0.66 inches/year) than the control plot. Runoff occurred only during snowmelt in March and April, periods of the year when runoff would be expected to have less impact on fisheries and macroinvertebrates due to the higher in-stream flows. The decrease in runoff was due to the reduction in the curve number observed between bare tailings (94) and the vegetated deep-plowed condition (65). Predicted annual soil loss for the control plot and deep plow treatment was 9.2 and 0.67 tons/acres, respectively.

6.2.2 Unsaturated Flow Modeling

The UNSAT2 model was used to evaluate the quantity and timing of percolation likely to occur in untreated and amended/revegetated tailings at three tailings sites: Rocker, Ramsay and Opportunity. Review of climatological data indicated that the risk of groundwater recharge varies seasonally. In mid-summer rainfall amounts are low and potential evaporation rates are high, hence the probability of percolation is small. Throughout fall and early winter, the precipitation and evaporation are nearly equal, but the soil profile is at least partially dry so minimal amounts of percolation would be expected. Starting with snowmelt and continuing through early summer, precipitation exceeds potential evaporation, hence percolation is most likely to occur in April through June.

UNSAT2 model runs simulated this climatic pattern. A domain was developed for each STARS site at Rocker, Ramsay, and Opportunity. Each soil profile was simulated for 1 year with the soil profile initially at "field capacity" (*i.e.*, the system was allowed to freely drain for 90 days). Next, a 90-day dry summer period (25 cm net water deficit) was simulated followed by a 180-day fall/winter period (5 cm of potential infiltration). Finally a 90-day snowmelt/late spring period was simulated where infiltration exceeded evaporation by up to 10 cm. A bare soil condition and a revegetated condition was simulated for each STARS site.

At the three sites, establishment of vegetation greatly increased the quantity of soil water use during the summer. The soil water deficit created was greater than the net infiltration the following winter and spring. As a consequence, no percolation was predicted from any of the reclaimed sites. For the unreclaimed tailings, the quantity of predicted percolation was 8 cm (3.1 inches) at Rocker, and 7 cm (2.8 inches) at Ramsay Flats (Figure 6.2.1.). Modeling difficulties at Opportunity made it more difficult to assess the net percolation rate, however the response at Opportunity appeared otherwise to be intermediate between Rocker and Ramsay.

SILVER BOW CREEK

Unsaturated Flow Model - Water Balance

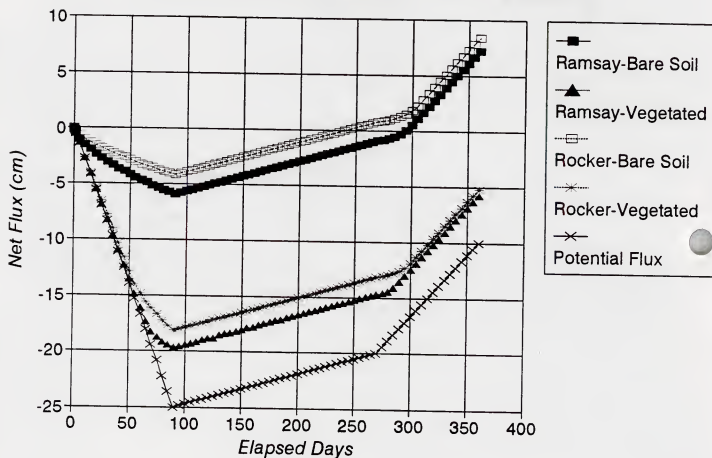


Figure 6.2.1. Simulation of cumulative net flux of water for vegetated and bare soil control plots at Ramsay Flats and Rocker. Positive values indicate groundwater recharge while negative values indicate net gain from groundwater by capillary rise.

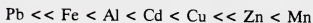
6.2.3 Geochemical Modeling

MINTEQA2, an EPA geochemical equilibrium model, was used to characterize the composition of pore water in untreated and treated tailings; simulate changes in metal solubility due to externally controlled increases in pH; investigate the effects of incremental limestone addition on pore water chemistry; and, evaluate arsenic behavior in the presence of an iron oxide adsorbate.

Pore Water Speciation: Increases in the pH of pore water samples due to lime addition resulted in a higher concentration of bicarbonate and hydroxyl ions relative to sulfate. Thus, at low pH, sulfate ion-pairing was common while at higher pH levels more metals were complexed with hydroxyl ion. Both lead and copper were paired with bicarbonate and carbonate at higher pH.

The saturation indices for various solid phase minerals indicate that iron and aluminum in pore water may have been at equilibrium with iron hydroxides and alunite, respectively. At higher pH levels, representative of amended soils, a number of precipitates were likely to form including copper as malachite ($\text{Cu}_2(\text{OH})_2\text{CO}_3$) and azurite ($\text{Cu}_3(\text{OH})_2(\text{CO}_3)_2$) and cadmium as octavite (CdCO_3).

Metal Solubility With Externally Fixed pH: When seepage containing metal-enriched pore water moves from a low pH tailings environment into underlying soil or sediment, the soil material being well-buffered will raise the pH of the solution. MINTEQA2 was used to simulate this external control of pH for several samples representative of tailings pore water. All metals precipitated at some threshold pH which was characteristic for each metal. Metals ranked from least to most soluble in the pH range of 5.5 to 7.5 were:



Measured metal concentrations were often well below levels predicted by MINTEQA2 especially at higher pH levels in amended STARS plots or in buried soil layers. As a consequence, it appears that other geochemical processes such as adsorption affect the behavior of copper, cadmium, and zinc at pH levels above 6.5.

Metal Solubility and pH Change With Incremental Limestone Addition: Addition of lime sequentially removed metals from solution. The order in which metals began to decline in solubility approximately followed the series described in the previous section. The pH response to limestone addition was strongly non-linear. Solution pH initially increased from approximately 3.7 to 5 with small additions of limestone. The system was briefly buffered around a pH of 4.7 to 5 due to precipitation of aluminum, iron and

copper. The pH again increased rapidly with additional limestone addition to a level of 6.7. Above this pH, approximately 80 percent of the total limestone added was required to increase the pH from 6.5 to 7.8. Formation of gypsum may have consumed the large quantity of limestone added in the simulation. The importance of the pH response to limestone is that zinc and manganese control, which occurs above a pH of 7, may be difficult using limestone.

Arsenic Behavior: Measured arsenic solubility in the field appeared to increase modestly (1 to 3 times) due to increases in pH from 3 to 8. This behavior could not be simulated by assuming solid phase control by ferric arsenate or calcium arsenate. Modeled arsenic concentrations were 1 to 5 orders of magnitude below the levels found in the presence of these solid phases. When arsenic was assumed to be controlled by adsorption on iron oxyhydroxide, then solubility was relatively constant across a wide pH range. Modeling also suggested that there the amount of adsorbent does not limit arsenic adsorption. Actual arsenic behavior may respond in part to both solid phase and absorption processes or competitive ion adsorption may favor adsorption of ions other than arsenic at higher pH.

At very low pH levels (below 3.2), very high arsenic was noted. Ion-pairing with various metal ions at low pH was proposed to explain this observation.

6.3 OVERALL TREATMENT PERFORMANCE

As discussed in the previous sections, each of the four main treatments (coversoil, agricultural tillage, deep tillage, and lime slurry injection) exhibited differences relative to each other for many of the investigative elements monitored. This section assesses those differences and the limitations of the treatments to mitigate metals contaminated tailings. The relative success of the STARS treatments to mitigate identified contaminant fate and transport mechanisms is also discussed.

6.3.1 Comparative Assessment of STARS Treatments

The main objective of the STARS treatability approach is to mitigate impacts from streambank tailings using chemical neutralization or isolation methods. The Phase I bench scale studies demonstrated the effectiveness of chemical amendments to reduce metal mobility by one or more orders of magnitude. The combination of chemical amendments which best reduced metal mobility were used in each of the treatments implemented at the study sites, although additional amendments (i.e phosphogypsum and ferric hydroxide) were added at some sites on some of the treatments.

Due to the heterogeneous nature of the STARS study sites and the fact that some monitoring data were not replicated, differences between different combinations of chemical amendments could not be discerned. Because of this, the primary differences in treatment effects that could be discerned between the different treatments (excluding the coversoil treatment) are based on the method used to mix the chemical amendments (agricultural tillage, deep tillage, and lime slurry injection). While there is no doubt that there are significant differences between treatments for some of the investigative elements presented in this report and summarized above, it is evident that there is no one best treatment identified in this study that is applicable to treating the variety of conditions encountered along Silver Bow Creek.

The results of the STARS study do show, however, that the two primary chemical amendment mixing methods, agricultural tillage and deep tillage, are each probably best suited for specific environmental conditions. It is also evident that lime slurry injection is not an efficient method of distributing lime to metal contaminated soils nor is it cost effective. The coversoil treatment, which may be the most protective treatment overall when considering human health concerns associated with direct contact, also has some direct advantages over any of the other treatments. Those environmental conditions that each method is most suited are presented in this section.

Agricultural Tillage Treatment: Agricultural tillage is most suitable for use in areas where metals contaminated soils are less than four to six inches (10 to 15 cm) thick due to the physical mixing limitations of the method. Agricultural tillage may also be useful in areas where groundwater is extremely shallow and deep tillage below the water table is not practical. Agricultural tillage methods can be readily adaptable to a variety of conditions and, qualitatively, agricultural tillage is the easiest treatment to implement. Agricultural tillage is the least expensive of the treatment methods evaluated on a cost per acre basis.

Deep Tillage: Deep tillage proved to be the most suitable method for mixing chemical amendments where the thickness of tailings was greater than 30 cm and less than 100 to 120 cm (3.5 to 4.0 feet). Mixing to this depth resulted in removal of sediment stratification and more homogenation of soils on deep till-treated plots. Incorporation of the amendments by this method was not only deeper than agricultural tillage but also had more uniform mixing at depth than the slurry injection method. Deep tillage is also effective in reducing total metals concentrations in the mixing zone due to the effect of dilution.

Although not as readily available as standard agricultural tillage equipment, several different types of equipment are commercially available. Drawbacks of deep tillage from an implementability standpoint include the higher horsepower requirements than the other treatment methods and the fact that deep plows are generally bulky, slow, and less maneuverable. Cost per acre for this method is approximately twice that of agricultural tillage.

Several other benefits were realized by using the deep plow. Deep tillage was effective at improving infiltration to considerable depths and causing a subsequent reduction in surface runoff. Vegetation production and cover were usually comparable to production and cover measured on the coversoil treatment, and generally higher than cover and production on agricultural tillage treated plots. One potential drawback of the method was the increase in concentrations of water soluble metals and As at the 120 cm depth interval. This result is likely due to the mixing of soils (or soluble salts) with higher concentrations of metals from the surface with soils containing lower concentrations of metals and As at depth. As a consequence, deep-plow treatment may not be suitable where groundwater is within 1 meter of the ground surface.

Coversoil Treatment: The coversoil treatment was the most effective when compared to other treatments in eliminating runoff and reducing erosion and exhibited the lowest concentrations of total metals at the surface. These results are consistent with the treatment method which is considered capping treatment than an amendment incorporation treatment. This treatment may be useful for amending small areas that are not amenable to treatment by other methods due to site specific limitations such as high water tables or poor soil characteristics, or at areas where direct contact exposure to surface soils is of human health concern. A drawback of cover-soil treatment is the lack of pH control within the tailings zone. Groundwater recharge through tailings areas treated by cover-soil application may contain higher metal levels than sites where tailings pH have been increased by amendment mixing.

Implementability of the treatment is similar to agricultural tillage because no specialized equipment is required. Because the treatment is actually an add-on to the agricultural tillage method, the cost to implement this treatment will always be more expensive than agricultural tillage. Costs expended to actually implement the treatment will depend on the availability of coversoil and the distance to the coversoil borrow source.

Lime Slurry Injection Treatment: The lime slurry injection treatment was able to incorporate amendments to depths of 4.0 feet or greater. However, mixing of the amendments was highly variable and generally inconsistent from plot to plot. The injected lime slurry tended to follow only the injection bores and moved into the soil profile only when large pores or fractures were encountered. Also, due to back pressure created when pumping the lime, much of the slurry was rejected, resulting in higher rates of lime ending up at the surface of the plot and less lime being incorporated at depth. Because of this, approximately only half of the lime needed to neutralize potential acidity could be applied to the injection plots.

Lime slurry injection is considered the least effective, least implementable and most expensive method of incorporation tested. Lime slurry injection was also found to be less effective than the other treatments in increasing pore water pH and the treatment was not

as effective at reducing runoff as the other methods of incorporation because excess lime discharged at the surface caused surface crusting.

6.3.2 Treatment Effectiveness in Addressing Contaminant Release Mechanisms

A conceptual model of the potential contaminant release mechanisms for metals in streambank tailings was described in Section 2.3. In this section, analytical and modeling results are used in combination to summarize the relative success of the STARS treatments to address identified contaminant fate and transport mechanisms.

Soil and Sediment Erosion to Surface Water: Although the STARS investigation did not directly measure or model direct sediment erosion to surface water, the STARS data and modeling support the well-documented phenomenon that vegetation cover provides considerable erosion protection.

The STARS rainfall simulation tests clearly demonstrated that the volume of rainfall runoff from tailings areas could be reduced considerably by establishing good vegetative cover on amended plots. The GLEAMS erosion model was used to estimate changes in the quantity of erosion that would occur when bare tailings in streambank tailings areas were revegetated using STARS techniques. Model results indicated that, for average rainfall years, STARS treatment would decrease erosion rates by 2 to 10 times. The reduction in erosion would be greatest in finer-textured tailings that have inherently higher erosion rates. As a consequence, STARS treatment is expected to greatly decrease the delivery of sediment to the stream due to sheet and rill erosion from streambank tailings areas.

The GLEAMS model also predicted that surface runoff from treated plots would generally be limited to the March through May period, presumably associated with snowmelt or spring runoff events. Stream flow during this period is typically high and the impacts from runoff and erosion are generally less detrimental to the stream during periods of high flow.

Runoff of Dissolved Metals to Surface Water: The movement of dissolved metals in runoff resulting from the dissolution of surface metal salts during convective thunderstorms is thought to be responsible for major fishkills where bare tailings areas are located near streams with fisheries. Dissolved metals in surface water generally are considered more bioavailable than other forms of metals and therefore have more pronounced impact on aquatic communities. Both the data and modeling from STARS support the conclusion that STARS treatments would be effective in reducing the level of dissolved metals loading to the stream.

The STARS rainfall simulation test data showed that metals concentrations in precipitation runoff from untreated tailings plots were well in excess of acute aquatic water quality criteria and contained a significantly higher percentage of dissolved metals when compared to runoff from treated plots. Treated, vegetated plots produced substantially less total runoff and the quality of runoff was substantially improved. The coversoil treatment essentially eliminated runoff at all sites and the deep plow treatment eliminated runoff at most sites.

The GLEAMS modeling predicted that most runoff on treated plots would occur during the months of March through May, whereas runoff from untreated plots would also occur during the summer. Any runoff occurring during the spring would have less impact on the aquatic environment of Silver Bow Creek because high stream flows during that period would provide a level of dilution and surface metal salt buildup typically is more significant later in the summer.

Leachate and Metal Loading to Groundwater: Three processes must occur in order for metals to move from a surficial tailings layer into groundwater. First, metals in the tailings layer must be soluble. Second, percolation must occur in order to convey the soluble metals downward. If the distance between the base of the tailings and groundwater is 30 cm or less, then diffusion (without convective movement of water) may account for movement of significant quantities of metals. Finally, the metals must remain soluble as they flow through the buried sediments in the vadose zone below the tailings.

Pore water and soluble metals determinations indicate that all streambank tailings sites contain highly elevated levels of copper, manganese, cadmium and zinc. Elevated iron and aluminum were common in those tailings deposits with a pH less than 4.0 to 4.5. Lead and arsenic were generally not highly soluble in the tailings. Addition of chemical amendments to the tailings greatly reduced the solubility of all metals. The degree of reduction in solubility accomplished by amendment addition varied by element. The relative reduction in metal ion solubility due to amendment addition from most to least reduction was:

$$\text{Al} \approx \text{Fe} > \text{Cu} >> \text{Cd} \approx \text{Pb} > \text{Zn} > \text{Mn} >> \text{As}$$

Arsenic solubility increased on some but not all amended plots. Its solubility should be considered when contemplating use of STARS techniques in areas where elevated total concentrations of As are encountered.

Soil water content monitoring data as well as vadose zone convective flow modeling indicated that untreated tailings would have 1 to 3 inches of groundwater recharge in an average rainfall year. Most of the percolation would be expected to occur in a 90 day period from early April until the end of June. Establishment of a healthy stand of vegetation through the use of STARS amendments was shown to greatly reduce the amount of recharge or possibly eliminate percolation entirely in an average year.

Pore water chemistry of buried sediments between the surficial tailings and the groundwater indicated that natural soil underlying the tailings has served as an effective geochemical barrier to the transport of metals at Ramsay, Fairmont, and Opportunity. At two sites (Manganese Stockpile and Rocker), "breakthrough" of metals into the groundwater system has occurred. The two findings are consistent with regional groundwater studies, which indicate that groundwater contamination is a localized rather than regional issue. The tendency for metals to reach groundwater appears to be controlled by the thickness and characteristics of the unsaturated zone between the base of the tailings and groundwater. Vertical separation of less than 50 cm (20 inches) between tailings and groundwater and presence of coarse-textured, weakly-buffered, slightly acid alluvial sediments, such as at Rocker, appear to have supported metal migration. The portions of Silver bow Creek where these conditions are likely to occur are the higher gradient reaches where the valley floor is narrow and confined by bedrock. the lateral extent of streambank tailings and the alluvial groundwater system in these reaches are confined to a narrow strip within tens of feet of the active stream channel. For streambank areas exhibiting these site conditions, the use of STARTS technology must be carefully weighed against other alternatives, taking into the account the importance of this pathway relative to others.

Fugitive Dust: No formal assessment of fugitive dust was completed as part of the STARTS investigation. Although fugitive dust migration from streambank tailings areas was noted on several occasions when field work was in progress, its importance as a metals migration route is uncertain. Nonetheless, establishment of vegetation on the tailings would greatly reduce the generation of fugitive dust from the tailings.

Bioaccumulation: Bioaccumulation of metals in vegetation constitutes a potential ecological and human health risk through their introduction into the food chain and ultimately, consumption by humans or other species. Analyses of major vegetation species during the STARTS study suggests that cadmium may be a potential problem at Rocker, Ramsay, and Opportunity, and that other metals may be of concern at Ramsay. Cadmium uptake by vegetation is controlled by the same plant physiological processes as the uptake of zinc, which is an essential plant micronutrient. Concentrations of cadmium often exceeded recommended tolerances for livestock consumption.

The STARTS investigation was not designed to definitively determine whether the elevated plant metals levels were due to uptake of metals into the plant tissue, deposition of metals onto the plants, or a combination of both mechanisms. Metal and arsenic concentrations for vegetation tissue from Ramsay were very high and within ranges that should have produced widespread plant morbidity. Such mortality did not occur. Therefore, the elevated levels were presumed to result from the effect of rainfall splash and dust accumulation rather than plant uptake. Although deposition of metals may still be of concern for bioaccumulation through grazing animals, such deposition should be greatly diminished through establishment of a good vegetative cover.

Direct Contact: Direct contact and incidental ingestion of the metals of concern, particularly arsenic, cadmium, and lead, are potential hazards associated with human exposure to streamside tailings. An implicit assumption of the STARS program has been that, in the rural nonresidential areas in which streambank tailings typically occur, the level of human exposure, and therefore potential risk, is limited.

Analysis of total metal levels at the surface of treated and untreated streambank tailings areas confirmed that most STARS treatments were not an effective means of reducing surface metal levels. Covering of tailings with a coversoil layer resulted in a decrease in surface metal levels as expected. Deep tillage at Fairmont, the site with the thinnest tailings layer, greatly reduced surface metal levels due to physical dilution.

Visual observation of the STARS plots indicated that the treatments were successful in reducing the formation of surface metal salt crusts. Elimination of those crusts, which have extremely elevated metals levels, is very important in reducing potential direct contact exposure.

It is evident also that establishment of a healthy vegetative cover on STARS treatments should significantly reduce the potential for direct human contact with surface soils. However, for those locations where significant exposure is projected, the coversoil treatment may be the only method that could effectively reduce surface metals to acceptable levels.

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